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CHEMICAL CHARACTERISTICS OF ONTARIO HYDRO COAL FLY ASH: A REVIEW

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Abstract

A literature review of the leachate characteristics of Ontario Hydro's coal fly ash was conducted to identify chemical parameters that may be environmentally significant in a landfill operation. Coal ash from Ontario Hydro's thermal generating stations burning bituminous and lignite coals were considered. Results from leach tests performed on the various fly ash samples confirm that Ontario Hydro fly ash can be classified as a non-hazardous, non-registerable waste in accordance with MOE Regulation 309. As a general trend, constituent concentrations in bituminous fly ash porewater were relatively higher than in lignite fly ash porewater. The potential environmental impact of major, minor and trace elements, as well as ionic complexes, which were detected in fly ash porewater, are discussed with respect to possible groundwater contamination.

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To Mr. D. K. Fung
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CHEMICAL CHARACTERISTICS OF ONTARIO HYDRO COAL FLY ASH: A REVIEW

1.0 INTRODUCTION

As part of the Lakeview Coal Ash Management Study, a review of laboratory and field data on the leachate characteristics of Ontario Hydro's coal fly ash was conducted to identify chemical parameters that may be environmentally significant in a landfill operation. Coal ash from three of Ontario Hydro's thermal generating stations burning bituminous coals were considered: Lakeview TGS, Nanticoke TGS, and Lambton TGS. In addition to these stations, the characteristics of fly ash derived from lignite coal burned at Atikokan TGS and Thunder Bay TGS were also included. For a description of the coal and ash handling system used at each station refer to Table 1 (Buckingham, 1989).

Results of the total (bulk) chemical composition, the composition of the Ministry of the Environment (MOE) distilled water and Regulation 309 leachate extracts, and porewaters are presented, when available, and discussed.

The total (bulk) chemical composition can be used to classify Ontario Hydro coal ash with the assistance of a compositional field diagram, while constituent concentrations within the leachate extracts, obtained as a result of the Ministry of the Environment (MOE) Regulation 309 leach test, are evaluated to classify fly ash in the appropriate waste category. Although MOE

laboratory leach test results are not representative of actual in situ leachate quality, porewater results do provide information regarding leachate chemistry generated under actual field conditions.

2.0 BULK CHEMICAL ANALYSIS

Bulk elemental analyses of samples of fly ash were obtained through a combination of neutron activation, a strong, mixed-acid digestion procedure followed by atomic adsorption spectrometry and plasma mass spectrometry methods. The results for both major and trace element constituents, given as μg/g, for ash samples derived from a U.S. bituminous coal (Lakeview), a blended U.S./W.Canadian coal (Nanticoke) and a lignite coal (Atikokan) are presented in Table 2 (Curtis, 1977; PEMRC, 1988 and Dayal and Zhou, 1989).

Based on the major element composition, Ontario Hydro fly ashes are predominantly composed of silica and alumina with smaller amounts of alkaline earth metals (Ca, Mg), alkaline metals (Na, K) and iron. If classified on the compositional field diagram proposed by Roy and Griffin (1982), the fly ash plots in the field designated as modic (see Figure 1). Since the leachate pH of the bituminous fly ash is alkaline, the classification can be prefixed to designate the pH character as alkaline modic. This classification scheme is graphically similar to that originally presented by Roy et al. (1981), however the nomenclature has been modified. Based on the Roy et al. scheme, a fly ash whose weight percent composition plotted in the 'modic' region would be termed ferrocalsialic. The lignite ash from Atikokan, although still classified as modic, has slightly lower sialic constituents (Si, Al, Ti) and higher quantities of the ferric and calcic group components than the bituminous or bituminous blended ash, and therefore plots in a slightly different location on the triangular diagram (Figure 1).

As seen from a comparison of Tables 2 and 3, the bulk chemical analysis of Ontario Hydro ash is well within the range of elemental concentrations reported for U.S. fly ash (Ainsworth and Rai, 1987). In fact, for the trace element analyses, Ontario Hydro ash is generally below the mean values for concentrations both on the total fly ash and when considering the bituminous and lignite subgroups individually. For example, the mean arsenic concentration for Ontario Hydro's ash is 89 μg/g, while Ainsworth and Rai present 156 μg/g as an average for U.S. coal ash.

As a general trend, fly ashes derived from bituminous coal burned at Lakeview and Nanticoke contain higher amounts of trace elements such as As, Se, Cr, V and Ti than the fly ash from lignite coal (Atikokan, Table 2). This trend has also been observed in U.S. fly ash as seen from Table 3.

2.1 Implications

In December, 1988, a program for an Improved Lakefill Quality Control Program (ILQCP) was introduced by the Metropolitan Toronto and Region Conservation Authority (MTRCA) to control the deposition of environmentally unsuitable materials at lakefill locations. To this end, a manual outlining the processes to be followed to assess the suitability of a fill material for lakefill has been produced (MTRCA, 1988).

The ILQCP policy outlines three categories of lakefill materials: suitable for open water disposal, confined disposal, and protected disposal. The categorization of each fill type is based on the flow chart shown in Figure 2. As seen from Figure 2, for the open water disposal, a bulk elemental analysis of the material must pass (i.e. be less than or equal to) the open water disposal guidelines (OWDG) shown in Table 4. These parameters are based on the maximum acceptable concentration of total sediment chemistry (i.e. total acid digestion). A comparison of Table 2 with the OWDG shows that (for parameters under strict control) mercury (Hg) concentrations exceed the guideline value of 0.3 μg/g in the "average" ash analyses (0.55 μg/g) reported by Curtis

(1977), cadmium (Cd) concentrations in both the Lakeview fly ash and "average" fly ash are over OWDG limits by a factor of two and arsenic (As) levels are exceeded by up to a factor of 10. In addition to these restricted parameters, the bulk elemental analysis in Table 2 shows that Ontario Hydro fly ash exceeds the recommended limits of several other of the constituents listed in Table 4. Therefore, based on the total elemental composition of Ontario Hydro fly ash, it cannot be used as an unrestricted lakefill material, i.e. it is not suitable for open water disposal as outlined in the ILQCP manual (MTRCA, 1988).

Comparison of the elemental composition of fly ash with the dredged material restricted land use clean-up criteria (Table 5), shows that Ontario Hydro coal ash exceeds these guidelines for As, Cr (total), Co, Fe, Mo, Ni, Se and V. Therefore, based on the MTRCA program, Ontario Hydro fly ash (unstabilized) could not be used as a lakefill material.

Proposed MOE lakefill policies, although slightly different than the MTRCA guidelines, are nevertheless just as restrictive with regard to classification of lakefill material (Persaud, 1989). Although only two categories for MOE lakefill material are suggested, that is, restricted or unrestricted fill, in order to be classified as acceptable for lakefill, the prospective fill material must pass the following set of initial screening criteria (Persaud, 1989);

i) total elemental concentration of material < clean-up guidelines for soils,

ii) $\leq$ 1x parameter values for MOE Regulation 309 Schedule 10 leach test, and

iii) $\leq$ 10x Ontario Provincial Water Quality Objectives and guidelines in a distilled water column leach test.

Let us now consider the first screening criteria. The parameter values for the clean-up guidelines for soils, as shown in Table 6, are dependent upon the proposed land use and grain-size or texture of the soil. Since fly ash is classified as a silty material, it would fall in the medium-to-fine textured

category of Table 6. Comparing the elemental ash concentrations of Table 2 with the commercial/industrial land use criteria (highest permissible levels), Ontario Hydro fly ash exceeds the guidelines for both As and Se (Table 2, column 6). When the preferred final land use is residential, parkland or agricultural, fly ash exceeds the recommended guidelines for As, Se and Mo.

The above comparisons indicate the unstabilized coal ash would likely not be considered as a suitable material for lakefilling, both by the MTRCA and MOE suggested guidelines. However, the MOE suggested initial screening criteria has not been finalized and until an official written document has been published, final considerations regarding the use of fly ash for lakefill cannot be made. It should also be noted that the addition of stabilizing agents or additives may alter the total elemental analyses sufficiently to reduce levels of As, Se and Mo to below guideline concentrations.

3.0 LEACH TEST RESULTS

3.1 MOE Regulation 309 Leach Tests

The Ministry of the Environment (MOE) developed the Regulation 309 Leachate Extraction Procedure as a method to determine whether a waste material is considered to be leachate toxic and, therefore, hazardous. The regulation states that a sample be prepared using a solid:liquid ratio of 1:20 and that the pH be adjusted to ~5 using acetic acid. The acetic acid is added in order to simulate co-disposal (with municipal refuse) landfill conditions where the generation of organic acids and microbial activity may lower the pH and hence, increase the solubility of heavy metals. After a 24 hour agitation period, an aliquot of the leachate is extracted and analyzed (MOE, 1985). Wastes are classified by comparing sample leachate extracts to leachate quality criteria outlined in Schedule 4 of Regulation 309. The concentration criteria in Schedule 4 are based on Ontario drinking water objectives for parameters related to health and are given in Table 7. Under the guidelines set forth in

Regulation 309 under the Environmental Protection Act (Part V), a waste which produces leachate containing any of the contaminants listed in Schedule 4 at a concentration in excess of one hundred times the value specified in the Schedule, is considered to be hazardous and is classified as a leachate toxic waste (MOE, 1985). Wastes which are classified as non-leachate toxic fall into two categories: registerable (leachate constituent concentrations 10 - 100 x Schedule 4 levels) and non-registerable (leachate constituent concentrations <10 x Schedule 4 levels). Refer to Table 7 for further clarification.

Results from the Reg. 309 leach tests performed on fly ash samples from Lakeview TGS, Nanticoke TGS, Thunder Bay TGS and Atikokan TGS, were summarized and evaluated to determine whether fly ash is considered, according to the MOE Reg. 309, to be a leachate toxic waste and, therefore, a hazardous material. The results are listed in Tables 8-11. Although all constituent concentrations fall well below the hazardous waste guidelines set by the MOE, some constituents are at levels which approach or exceed values listed in Schedule 4. A visual representation of these constituents, displayed as a percent ratio of the associated maximum acceptable concentration, is presented for each station in Figures 3-6, where:

$$\text{percent ratio} = \frac{\text{concentration of solute in leachate extract}}{100 \times \text{solute concentration in Schedule 4}} \quad (1)$$

In Lakeview fly ash leachate, six constituents were present at levels above those listed in Schedule 4 (Figure 3). Cadmium had the highest percent ratio (7%) in the Lakeview extract, followed by selenium (5%), boron (3%), chromium (2.2%), uranium (1.5%) and arsenic (1%). Nanticoke fly ash leachate contained five constituents which approached or exceeded the Schedule 4 values. In this case, selenium had the highest percent ratio, a value of 9%, followed by barium which was present at a 5% ratio. Boron, arsenic and uranium, which were also detected in the Lakeview fly ash leachate, were present in Nanticoke leachate with percent ratios of 4.7%, 3.3% and 2.4%, respectively

(Figure 4). From Figure 5, which represents Thunder Bay fly ash leachate, it is apparent that of the four constituents depicted in this graph, selenium had the highest percent ratio, calculated to be 10%.

The other three constituents represented are boron (4.2%), cadmium (4%) and lead (2%). Atikokan fly ash leachate contained the greatest number of constituents which were in excess of the values listed in Schedule 4, however, all were present at levels of less than ten times the Schedule 4 concentrations. Selenium had the highest percent ratio at 6.2% (0.062 mg/L), succeeded by boron (5.8%), uranium (3%), chromium (1.2%), cadmium (0.9%) and fluoride (0.9%) (Figure 6).

It is interesting to note some of the significant features of the four graphs. For example, fly ash samples, from all four stations, produced leachates containing concentrations of selenium, boron, uranium and cadmium in excess of values listed in Schedule 4. However, even the constituent exhibiting the highest percent ratio, selenium, is only within 10% of the hazardous waste guideline (maximum observed concentration of 0.1 mg/L from Thunder Bay fly ash leachate). Leachate derived from Atikokan fly ash, gave the highest concentrations of both boron (29 mg/L, 5.8% ratio) and uranium (0.06 mg/L, 3% ratio), while cadmium levels were found to be greatest (0.035 mg/L, 7% ratio) in the Lakeview fly ash extracts. Elevated concentrations of arsenic were present in all of the bituminous ash leachates, however, the Nanticoke samples contained the uppermost level of 0.166 mg/L (representative of 3.3% of the maximum acceptable concentration). The highest concentration of chromium was found within the Lakeview fly ash leachates (0.11 mg/L, 2.2% ratio). Other results of note are the maximum leachate concentrations for fluoride (2.2 mg/L, 0.9% ratio) from the Atikokan fly ash, barium (5 mg/L, 5% ratio) from Nanticoke fly ash and lead (<0.1 mg/L, 2% ratio) from Thunder Bay fly ash.

On the basis of these results, Ontario Hydro fly ash would be classified as a non-hazardous (non-registerable, non-leachate toxic) waste in accordance with Regulation 309.

3.2 MOE Distilled Water Leach Tests

The MOE distilled water leach test is a laboratory procedure used to assess the leaching behaviour of a material where there is no potential for an acidic leachate to be generated (MOE, 1983b). The method is similar to the Reg. 309 Leachate Extraction Procedure, however, the pH of the mixture is not adjusted.

Results from distilled water leach tests performed on fly ashes from Lakeview TGS, Nanticoke TGS, Thunder Bay TGS and Lambton TGS are summarized in Tables 8 to 10 and Table 12. The pH measurements of the distilled water fly ash leachates ranged from 9.5 to 11.2, an alkaline solution, which is significantly different than the acidic nature of the Reg. 309 leachate extracts which had pH values ranging from 4.9 to 5.2. As was noted for the Reg. 309 results, selenium and boron were detected at levels exceeding the Schedule 4 recommended limits of 0.01 mg/L and 5 mg/L (respectively) in leachates from all four stations. Thunder Bay fly ash leachate contained the maximum levels of these two constituents with a selenium concentration of 0.12 mg/L and a boron concentration of 7.5 mg/L. Generally, however, the distilled water extracts had relatively lower constituent concentrations than did the Reg. 309 leachates, likely due to the fact that alkaline conditions normally inhibit the release of trace metals. With this in mind, while co-disposal is an acceptable option, a monofill landfill site would be preferred for the disposal of Ontario Hydro fly ash, in order to reduce the likelihood of acid generation and microbial activity associated with municipal refuse that would, in turn, increase trace element leachability.

4.0 POREWATER CHEMISTRY

Porewater (in situ leachate) associated with coal ash provides the means to characterize leachate generated under field conditions and is of concern due to the possible environmental impact of trace elements and ionic complexes which may enter local or regional groundwater regimes. Fly ash porewater

samples were obtained from piezometers with well screens located within the ash piles at Lambton, Nanticoke and Atikokan TGS. The results of the chemical analyses of these in situ leachates are given in Tables 14 through 16, respectively. Table 13 contains porewater chemistries of Lakeview fly ash which are a combination of leachate analyses for porewaters obtained from both piezometer data (Birchwood Park) and interstitial waters extracted from undisturbed ash cores taken from Birchwood Park. Tables 13 to 16 include both the mean solute concentrations as well as the overall range of concentrations observed.

The porewater analyses were compared to the Ministry of the Environment (MOE) Ontario drinking water objectives and guidelines (PWQO/G) given in Table 17 (MOE, 1983), to identify solutes which should be targeted as requiring further investigation with respect to possible groundwater contamination.

4.1 Major Constituents

The bituminous fly ash porewaters generated from the dry landfill situations (Lakeview and Lambton) exhibit alkaline pH's and elevated concentrations of Ca^{2+} and SO_4^{2-}. In fact, the sulphate concentrations generated in the ash leachate exceed the PWQO/G of 500 mg/L by factors of 3 to 4 (average concentrations of 1743 mg/L and 2007 mg/L for Lambton and Lakeview respectively), and would therefore be considered as a potential contaminant.

The sulphate and calcium results from the Nanticoke ash porewaters seem anomalously low for a bituminous derived coal ash (mean values for Ca^{2+} and SO_4^{2-} of 141 mg/L and 294 mg/L respectively) and are likely due to the combination of coals used and the ash handling technique employed (ash is sluiced and subsequently dredged for disposal). It is anticipated that if Nanticoke TGS were converted to a dry landfill operation, the concentrations of major ions in the ash porewaters would be higher.

Average sodium concentrations in porewater from Lakeview, Lambton and Nanticoke were determined to be 350 mg/L, 92 mg/L and 53 mg/L (respectively). As Na is not considered to be a toxic metal, a maximum acceptable concentration for drinking water has not been set by the Ministry of the Environment, and it therefore is not likely to be considered a contaminant. However, they do recommend that Na concentrations in excess of 20 mg/L be reported to the Ministry of Health for notification to persons on a Na restricted diet if the water is to be used for domestic supply.

In contrast to the bituminous fly ash results, porewaters obtained from the lignite-derived ash at Atikokan do not exhibit major ion concentrations in excess of PWQO/G and therefore would not be of environmental significance.

4.2 Minor/Trace Constituents

With respect to the minor and trace constituents of the fly ash porewaters, depending on the station, concentrations of boron, selenium, arsenic and, to a lesser extent, chromium, manganese and phenol have been detected at levels in excess of the PWQO/G. The potential environmental impact of each of these elements is discussed below.

4.2.1 Boron:

Boron values were found to vary widely, and for the bituminous coal ash leachates, often exceed the MOE PWQO/G limit of 5 mg/L. For example, boron values in Lakeview fly ash porewaters vary from 17.8 to 51 mg/L with a mean concentration of 28.6 mg/L. This exceeds the drinking water objectives by a factor of ~6. Unlike the porewater generated by bituminous fly ash, in situ leachates from lignite derived fly ash were less than 0.1 mg/L (Table 16) and is therefore unlikely to be of environmental concern. For bituminous coal ash, however, boron should be identified as a potential contaminant.

Being a potential contaminant, the long-term leaching behaviour of boron is an important factor to be considered. A column experiment was therefore conducted by Chan and Johnston (1983) to evaluate the release characteristics of selected ions from a sample of Lakeview fly ash obtained from the Birchwood Park landfill site. The weathered ash was placed in a series of four columns and continuously leached with deionized water. The chemical characteristics of the ash, for boron, versus the appropriate cumulative pore volumes are shown in Figure 7. The experiment was subsequently repeated using a much smaller column system with similar results (Johnston, unpub. results; see Figure 7). Initial characteristics of the leachate include maximum concentrations of boron (27 mg/L) as well as other major ions. Over time, however, the boron concentration (as well as Mg, Na, K and Cl) decreased significantly. As seen in Figure 7, the boron content of the leachate declined to below the Ontario drinking water objective of 5 mg/L after being eluted with less than 10 pore volumes of deionized water.

Warren and Dudas (1986) also noted that boron is readily dissolved from fly ash and Choi and Chen (1979) further suggest that the high solute concentrations observed in the initial pore volumes increase boron solubility, thus enhancing leachability. The rapid depletion of boron from the ash indicates that it is primarily associated with the reactive surface area of the ash particles. Warren and Dudas (1986) suggest that boron is present as an oxide such as B_2O_3 which hydrolyzes to form $H_3BO_3^{\circ}$ and its subsequent dissociation products.

Although it is not the purpose of this report to discuss anticipated migration rates of various solutes found in the ash leachate, it is important to note that these solutes may have limited mobility due to uptake reactions with geologic materials (i.e. contaminant migration rates may be less than the average linear groundwater velocity within the disposal area).

The form of boron in solution is important with respect to the uptake of B species by geologic minerals. Boron in aqueous solutions exists primarily as boric acid ($H_3BO_3^o$) and, under highly alkaline conditions, $B(OH)_4^-$; the dissociation constant being

$$\frac{[H^+]\ [B(OH)_4^-]}{[H_3BO_3^\circ]} = 10^{-9.23} \quad (2)$$

Therefore, for pH's above 9.23, as would occur in fresh ash leachate, $B(OH)_4^-$ is the dominant species. As the leachate migrates away from the landfill/disposal area, it reacts with the geologic medium and mixes with the local groundwater, thereby decreasing the pH. With the pH decreasing to less alkaline values, boron will be primarily in the form of the neutral species $H_3BO_3^o$.

In the uptake of contaminants by overburden materials, the transfer of mass from the porewater to the solid phase of the porous medium causes the rate of advance of the contaminant front to be retarded with respect to groundwater flow velocities. For chemical species that undergo transfer between the liquid and solid phases as a result of ion exchange (adsorption-desorption processes), the equilibrium partitioning of the species between the two phases is commonly described by a distribution coefficient (Kd) where:

$$Kd = \frac{\text{mass of solute on the solid phase per unit mass of solid phase}}{\text{concentration of solute in solution}} \quad (3)$$

The dimensions of the distribution coefficient is therefore L^3/M and is commonly reported in units of mL/g.

Keren and O'Connor (1982) investigated the boron adsorption capacity of selected clay minerals. Using an ionic strength of 0.02 they determined distribution coefficients of ~ 2-4 mL/g for Na-montmorillonite and ~6 mL/g for Ca-montmorillonite at a pH of 9. In similar experiments using illite they determined Kd values of between 13 to 16 mL/g. As seen from their results, Na-illite exhibited a much higher capacity for B adsorption than Na-montmorillonite. Sims and Bingham (1967) determined that maximum boron

retention for Na-vermiculite occurred between pH values of 10.5 and 11 while the Na, K-kaolinite maxima occurred at ~pH 9. Sims and Bingham also determined that B retention by montmorillonite was slight compared to kaolin and suggested that the presence of hydroxyl Fe and Al, present as interlayer materials/impurities played a dominant role in boron uptake.

Although anionic species are typically sorbed to a greater extent than neutral species, even the neutral $H_2BO_3^o$ can be sorbed to some degree depending on the mineralogy of the overburden (Sakata, 1987). For example, Sims and Bingham (1968a) determined the maximum uptake of boron by hydroxyl iron materials (sesquioxides) occurred at a pH of 8.5. Similar experiments conducted on hydroxyl Al materials showed that retention was ten times greater than measured in the hydroxyl Fe experiments with maximum adsorption at a pH of ~7. Further work on boron uptake by Fe and Al-coated clays showed that boron uptake was highly pH dependent with maximum retention occurring between pH 8 and 9 (Sims and Bingham, 1968b). The results showed a high correlation between boron retention and the free iron oxide content of the soils studied while only a fair relationship between B adsorption and Al oxide content was detected. Investigations by Keren and Gast (1983) support Sims and Bingham's findings. They observed that colloidal clay and Al and Fe oxides were the main components for B adsorption by soils. In addition, Okazaki and Chao (1968) noted that boron adsorption was not completely reversible, suggesting that chemi-sorption or some other permanent fixation of B onto soils also occurs.

Although the above discussion illustrates that some adsorption of boron is possible and indeed is likely to occur, boron is still generally considered to be potentially mobile in groundwater environments and, since it exceeds the PWQO/G by up to an order of magnitude (in porewaters from bituminous-derived fly ash), it should be regarded as a primary contaminant of potential environmental concern.

4.2.2 Selenium:

Selenium concentrations at or above the PWQO/G value of 0.01 mg/L were detected in porewater samples obtained from Lakeview, Lambton and Nanticoke fly ashes. The lignite-derived ash samples showed no measurable Se content i.e. porewater Se concentrations were below the detection limit of 0.001 mg/L. Of the porewaters collected from the bituminous coal ashes, Lakeview ash exhibited the highest measured concentrations with all five samples showing Se concentrations above 0.01 mg/L (Se levels ranged from 0.012 to 0.052 mg/L) with an average concentration calculated at 0.028 mg/L.

Of the twelve samples analyzed from Nanticoke TGS, four were found to have Se levels above the PWQO/G. Se concentrations at Nanticoke varied from below detection (0.002 mg/L) to a high of 0.028 mg/L. On average, the selenium concentration of the porewaters were equal to the drinking water objective of 0.01 mg/L.

A similar average concentration for Se was determined for Lambton porewaters. Although only three samples were obtained from this site, only one (0.013 mg/L) showed Se concentrations in excess of the PWQO/G. The remaining samples had Se levels of 0.008 mg/L and 0.009 mg/L.

Due to the elevated selenium concentration observed in porewaters obtained from Lakeview TGS fly ash, and the fact that both Nanticoke and Lambton showed some porewater samples with Se concentrations exceeding the PWQO/G of 0.01 mg/L, Se should be considered as a primary contaminant.

With respect to the long-term leaching behaviour of selenium, column experiments (as described in section 4.2.1) were conducted on samples of Lakeview fly ash obtained from Birchwood Park (Johnston, unpub. results). The results, as shown in Figure 8, differ significantly from those of boron (Figure 7). Whereas the initial characteristics of the boron leachate profile exhibited maximum B concentrations (27 mg/L) which decreased rapidly to below

the PWQO/G within 10 pore volumes (PV), the initial Se concentration of 0.024 mg/L was not the highest attained. Rather, selenium levels increased to a maximum concentration of 0.037 mg/L at 50 PV before levels began to decline and selenium concentrations did not decrease to below the PWQO/G of 0.01 mg/L until 175 PV of deionized water had been passed through the column.

The results of the column experiments indicate that, unlike boron, the majority of Se is not present as readily leachable surface oxides on the fly ash particles since Se oxides are extremely soluble and would be rapidly dissolved on contact with water. Indeed, Warren and Dudas (1986), in long-term (770 PV) column experiments conducted on coal fly ash, noted that, in general, only a portion of Se is present as an easily hydrolyzable form on the ash particle surface. Andren and Klein (1975) determined that the majority of Se was precipitated on the surface in its elemental form (Se^{o}) which, in an oxidizing environment, would be oxidized to the +4 (selenite) and +6 (selenate) states. Rai et al. (1987) also suggest the possibility of a $FeSe_2$ solubility control while Warren and Dudas (1986) suggest that the majority of Se in fly ash is actually incorporated into the exterior reactive glass of the ash particles. At present, however, the exact nature of the solubility control for Se release (leachability) does not appear to be accurately known (Rai et al. 1987). What is evident from the column leaching experiment is that Se levels will increase over time but will not persist in perpetuity.

The mobility of Se in the subsurface is affected by its chemical properties which are controlled by its oxidation state. Selenium exists in the natural environment in four oxidation states: 0, -2, +4 and +6, however, dissolved Se species are predominantly in the form of selenites (Se(IV)) and selenates (Se(VI)). Research conducted by van der Sloot et al., (1985) has shown that the major form of Se in coal ash leachate is Se(IV). From Figure 9, the dominant forms of Se(IV) in solution are $HSeO_3^-$ for pH values below 8.27 and

With respect to the potential mobility in the subsurface, it should be noted that although Se is anionic in nature, sorption onto geologic material can be significant (Griffin et al., 1977). For example, although the net electrostatic charge of clay particles is normally negative and therefore attracts cationic species, there is also a small component of positively charged sites. The major anion adsorption sites on clay minerals are associated with broken bonds on particle edges in which the octahedral sheet is broken, exposing positive charges (van Olphen, 1977). In addition, the $HSeO_3^-$ ion appears to be the Se(IV) species predominantly adsorbed by clay particles (Frost and Griffin, 1977).

Singh et al. (1981) noted that, in addition to the clay content, the organic content, cation-exchange capacity (CEC) and the amount of calcite present were also important in the adsorption of selenite species by soil. Sorption/desorption experiments of Se on various minerals conducted by Ticknor et al. (1988) at a pH of 7.5 indicate that some phyllosilicates (chlorite, illite and kaolinite) are effective in removing Se (predominantly $HSeO_3^-$) from solution under both aerobic and anaerobic conditions, with sorption under aerobic conditions being irreversible. Since S. Ontario clays are generally illitic in nature, Se adsorption could prove to be significant. For example, laboratory batch experiments conducted on samples from the St. Clair Clay Plain from the Lambton TGS site showed average Se Kd values to be ~16 mL/g (Johnston, 1982a).

In addition to uptake of Se by clay minerals, Ticknor et al. (1988) noted that several other minerals had the capacity to adsorb Se as shown in Table 18. Of special note is the capability of gypsum to remove both $HSeO_3^-$ and SeO_3^{-2} from solution. The mechanism suggested for this was exchange with SO_4^{-2}. Since gypsum is commonly found as a fracture fill material in the St. Clair clay plain, this could prove to be a significant retardant to Se mobility in the upper fractured clay zone in this area. Further, as was also found for boron, iron oxides/hydroxides are also recognized as efficient scavengers for anionic trace elements such as Se (Ticknor et al., 1988; Fritz and Hall, 1988; Sakata,

1987; Elkhatib et al., 1984) and are common components of the upper weathered zone in unconsolidated materials. For example, laboratory studies of the potential mobility of Se in the sandy overburden at the Canadian Armed Forces Base Borden have shown Se to be retarded by a factor of 3 to 4 relative to groundwater flow velocities (Johnston, unpub. results). Even gabbro, basalt, granite and calcite have been shown to have some affinity for selenium uptake (see Table 18).

Although selenium concentrations in ash porewaters are above the PWQO/G and are likely to remain elevated over an extended period of time, the above discussions illustrate that Se adsorption can be a significant retardation mechanism and suggest the relative immobility of Se in most groundwater environments.

4.2.3 Arsenic:

As seen from Tables 13 to 16, arsenic concentrations in the fly ash porewaters vary from below the detection limit of 0.001 mg/L to concentrations in excess of the PWQO/G value of 0.05 mg/L, depending on the type of coal used and the ash handling procedure employed. For example, all 35 samples of porewater analyzed from the lignite-derived coal ash at Atikokan (Table 16) had As concentrations at or below 0.001 mg/L, with an average concentration of <0.001 mg/L.

A similar statement can be made for the Nanticoke ash. Although the mean concentration was over an order of magnitude higher than that observed for the Atikokan porewaters (i.e. 0.012 mg/L versus <0.001 mg/L), of the 12 samples analyzed, none were found to have As levels above 0.05 mg/L (Table 12). In fact, the highest value (0.025 mg/L) was only half of the PWQO/G. These relatively low arsenic concentrations may be a function of the wet handling system currently being used at Nanticoke. As was the case for the major ions Ca^{+2} and SO_4^{2-} the As levels in the associated ash porewaters may increase if a dry landfill system were employed.

Arsenic concentrations varied from 0.012 to 0.055 mg/L from the Lambton ash porewaters. Although the calculated mean of 0.037 mg/L shown in Table 10 is below the PWQO/G value of 0.05 mg/L, this estimate is based on only three samples and is therefore unlikely to be representative, especially in light of the fact that one of the values measured was above the PWQO/G (0.055 mg/L) and another was slightly below this value (0.045 mg/L). The range in As concentrations from the Lakeview ash porewaters was 0.012 to 0.067 mg/L with a mean of 0.044 mg/L. However, of the 5 samples analyzed, three were above the PWQO/G (0.052 mg/L, 0.06 mg/L and 0.067 mg/L). Based on these concentrations, coupled with the Lambton fly ash results, arsenic should be regarded as a primary contaminant, particularly in light of the column experiment results discussed below.

Results of a long-term, column-leaching experiment on a sample of Lakeview fly ash (as described previously) are shown in Figure 10 (Johnston, unpub. results). The initial leachate concentration observed was 0.024 mg/L at 0.68 PV which is within the range of values measured in Lakeview ash porewaters (Table 13). The concentration in the effluent began to increase significantly over time, however, reaching a maximum value of 0.39 mg/L at ~101 PV. After ~660 PV, arsenic concentrations approched the PWQO/G value of 0.05 mg/L but did not decline below this limit until over 1100 pore volumes of deionized water had been flushed through the system. This trend is again in contrast with the boron column results and even differs from the selenium leachate profile in that, although the shape of the curve is similar, the volume of water required to lower solute levels to below the acceptable drinking water limit was a factor of 6 larger. As was the case with Se, arsenic levels will persist at concentrations in excess of PWQO/G values for an extended period of time before eventually declining to levels no longer considered an environmental threat to drinking water supplies.

The results of the column experiment for arsenic suggest that As is associated with the exterior surface of the ash particle, however, the actual phases present have not been adequately studied and, hence, are not well known. For

example, Theis and Wirth (1977) suggested that arsenic was associated with Fe metal oxides while Hulett et al. (1980) determined that As was associated with the glassy matrix, but they did not differentiate possible surface precipitates from matrix associated forms. From results of column experiments, Warren and Dudas (1986) also speculated that arsenic was incorporated into the exterior glassy surface of the ash particles. van der Sloot and Niewendijk (1985) separated fly ash into two fractions, magnetic and glass, prior to acid leaching tests. They found As to be surface enriched in the Fe fraction, similar to Hulett et al. (1980), but also noted enrichment of As in the glass fractions. Since the form of solid As phases in the fly ash controls its release into solution, research in this area is greatly needed.

Arsenic exists in most natural groundwaters in either the +3 or +5 oxidation states. As with any multivalent solute, the speciation of arsenic in solution is a dominant factor in its relative mobility, and is therefore of prime importance. There is still uncertainty, however, as to the speciation of arsenic leached from fly ash. Turner (1981) determined that As is initially released into leachate as As(III) (due to dissolution of As_2O_3). Hansen et al. (1984), however, found As(III) and As(V) to be in approximately equal quantities, while Silberman and Harris (1984) observed As(V) to be the dominant species leached into solution. Results of research conducted by van der Sloot et al. (1985) confirmed Turner's observations while column tests performed by Warren and Dudas (1986) determined that As(V) was the dominant species initially leached from fly ash particles. Sato (1986), using batch leaching techniques, also found the dominant form of arsenic to be As(V). Regardless of the initial form of arsenic in solution, however, under oxidizing conditions As(III) will readily be oxidized to As(V), the most stable form in non-reducing environments. Laboratory studies by Cherry et al. (1979), for example, showed that even small amounts of As(III) are oxidized to As(V) by dissolved oxygen within a few days.

From the above discussions it is thus anticipated that As(V) would be the predominant species in fly ash leachate as it migrates away from a disposal area and discussions on the relative mobility of arsenic given below will therefore focus on the +5 (arsenate) oxidation state. In general however, under non-acidic conditions, adsorption and precipitation reactions are highly significant for both the +3 and +5 oxidation states, hence, arsenic is not considered to be very mobile in groundwater systems, as the following discussion will illustrate.

As seen from the distribution of hydrolysis products shown in Figure 11, in the pH range from 7 to 11.5, the dominant form of As(V) in solution would be $HAsO_4^{-2}$, while below a pH of 7, $H_2AsO_4^-$ would be dominant; both forms being anionic in nature. Arsenic (V) in the form of $HAsO_4^{-2}$ was shown to be the dominant arsenic species adsorbed by clay minerals with montmorillonite adsorbing more As (both +3 and +5) than kaolinite (Frost and Griffin, 1977). Frost and Griffin suggested the increased sorption by montmorillonite was likely due to the larger edge surface area. Maximum As(V) uptake by clay minerals occurred in the pH range of 4 to 6 (comparable to the $H_2AsO_4^-$ stability field in Figure 11) while similar experiments using As(III) showed increasing adsorption of As^{+3} as the pH rose from 3 to 9.

Arsenic, in general, has been shown to be significantly adsorbed by oxides (Fe, Mn, Al). Leckie et al. (1980) demonstrated that arsenate is very strongly sorbed onto am-$Fe(OH)_3$ with over 95% removal below a pH of 9.5. Arsenic was still observed to be strongly removed by am-$Fe(OH)_3$ above a pH of 9.5, however, in such alkaline solutions precipitation reactions, as well as sorption, were suggested to occur. In column experiments by Warren and Dudas (1986), for example, enrichment of As was found in the lower section of the column which indicated adsorption of arsenic on precipitated Fe oxyhydroxides. They further concluded that under moderately alkaline leaching conditions (pH < 9), arsenic was not mobile. The immobility of arsenic was also attributed to the formation of carbonates within the weathering ash which in turn reduced

suggested that the leached arsenic was removed by co-precipitation, with AsO_4^{3-} being incorporated in the calcium carbonate matrix. van der Sloot and Nieuwendijk (1985), from their leaching experiments, also suggest that arsenic is retained by adsorption or precipitation immediately after release into solution. Gulens et al. (1979) conducted column studies to establish arsenic mobility in sandy soils. Their results showed that, in a fine-grained sand at a pH of 5.7, As(V) was sorbed a factor of 8 times more than As(III) in an oxidizing environment while in a reducing environment, both species underwent little adsorption. When a medium-grained sand containing both Fe and Mn oxides (0.8% and 0.13% w/w, respectively) was employed, adsorption of both As species increased. They attributed the strong retention of As(V) in the sand to adsorption by hydrated ferric oxide. Although less adsorption was noted for As(III) than was found for As(V), it was anticipated that adsorption would increase with increasing pH, as was found by Griffin and Frost (1977). The ability of a sandy soil to sorb arsenic was also shown in laboratory uptake experiments conducted on samples taken from the Canadian Armed Forces Base Borden. Distribution coefficients (Kd) of 35 to 75 mL/g for As(V) were measured using groundwater obtained from the site and solid solution ratios typical of in situ values (Johnston, unpubl. results).

With respect to field studies, the combined laboratory/field investigation of the Lambton TGS ash disposal site is a good example of the immobility of arsenic species in a clay environment (Johnston et al. 1985). Laboratory measured As(III) Kd values in excess of 1500 mL/g were obtained using clay and groundwater samples from the Lambton site. Using these values, predicted concentration profiles for As beneath the fly ash disposal area at Lambton showed excellent agreement with the actual in situ As distribution. The field results showed that, after a period of 16 years, arsenic had migrated a distance of only 2.7 cm. Even in a sandy soil, adsorption of As has been shown to be significant. Killey and Myrand (1985) studied the sediments and porewaters from contaminated soil and the underlying sediment at the waste management area F at the Chalk River Nuclear Laboratories. They observed that although a small portion of the arsenic had been leached from the contaminated

soil, it was almost completely sorbed by the hydrous iron-oxide coatings on the underlying soil. Killey and Myrand measured the maximum porewater concentration of As within the contaminated waste to be 0.2 µg/mL yet, 4 to 5 m below the waste, porewater As concentrations in the interstratified sand and silt had been reduced to an average value of 0.009 µg/mL with the background concentration in the area being 0.007 µg/mL. Sorption had therefore reduced the dissolved arsenic concentration to near-background levels.

The above discussions illustrate the relative immobility of arsenic under both laboratory and actual field conditions. Given the proper disposal scenario (either engineered or natural), and considering the concentrations and leachate life-span of arsenic in Ontario Hydro fly ash, arsenic should be relatively immobile and not pose a significant environmental concern.

4.2.4 Chromium:

Chromium was detected at levels above the PWQO/G of 0.05 mg/L in some porewater samples obtained from Atikokan and Lakeview fly ashes. Cr in Lakeview porewaters ranged from <0.02 mg/L to 0.2 mg/L with a mean value of 0.08 mg/L. However, of the five samples of porewater analyzed, only one had Cr concentrations in excess of the PWQO/G (0.2 mg/L). The other four samples were below the 0.05 mg/L drinking water objective. Given the fact that Cr values in excess of PWQO/G were not detected in porewaters from other bituminous-derived fly ashes, the 0.2 mg/L value is considered anomalous. Due to the small sample size, the relatively high concentration level of 0.2 mg/L would influence the overall mean for the samples significantly and, although included in the calculation given in Table 13, it is felt that the average concentration of 0.08 mg/L is misleading and the actual mean is below the PWQO/G value of 0.05 mg/L.

The range of Cr detected in Atikokan porewaters was also large, varying from <0.05 to 0.1 mg/L with a mean of <0.05 mg/L (i.e. below PWQO/G). However, in this case the average is based on 35 samples with one value equal to 0.05 mg/L and only two of the remaining 34 samples showing Cr concentrations in excess of the PWQO/G (both at 0.1 mg/L). In this case the large sample size gives a more reliable estimate of the overall average Cr concentration in porewaters at the Atikokan TGS site.

From the above discussion it is therefore felt that although detected at levels in excess of 0.05 mg/L in a few samples, these values are considered anomalous and Cr is not a potential contaminant from Ontario Hydro fly ash. Nevertheless, some discussion concerning the relative mobility of Cr is presented below.

The PWQO/G for Cr is given as 0.05 mg/L. This value refers to Cr in the 6^+ oxidation state (Cr(VI)). The values given in Tables 13 through 16, however, are for total Cr, that is, both Cr(III) and Cr(VI). It should be noted that Cr(III) is the most common naturally occurring oxidation state for chromium and in this form it is not considered to be toxic. Harmful health effects are attributed to the hexavalent (VI) form which is stable under strong oxidizing conditions. It is believed that, due to the oxidizing environment under which fly ash is formed, Cr likely occurs in fresh ash primarily in the Cr(VI) state (Warren and Dudas, 1986).

However, speciation of Cr, as with B, is affected by soil/solution pH. For Cr(VI), for example, above a pH of 6.4, $HCrO_4^-$ dissociates to CrO_4^{2-} which then becomes the dominant form. With respect to Cr(VI), Warren and Dudas (1986), in studies of trace element migration through fly ash columns found Cr to be mobile under alkaline conditions. Similar research conducted by Leckie et al. (1980) on chromate (CrO_4^{2-}) adsorption onto amorphous iron hydroxide, determined that the percent CrO_4^{2-} adsorbed decreased from >90% to near zero

over the pH range of 6 to 8. Therefore, in the normal pH range typical of S. Ontario soils (7 to 8), Cr(VI) is likely to be mobile, exhibiting migration rates at or near groundwater velocities.

Solution pH also affects the rate of reduction of Cr(VI) to Cr(III) with lower soil pH favouring rapid reduction (Bartlett and James, 1979). However, even at the pH values and redox potentials of most soils, Cr(VI) is likely to be readily reduced to Cr(III) (de Haan and Zwerman, 1978). Reduced species are strongly sorbed by Al and Fe compounds and would therefore migrate at rates substantially lower than that of groundwater and, coupled with the fact that Cr(III) is not considered to be toxic, it would not be of environmental concern.

4.2.5 Manganese:

Elevated concentrations of manganese were detected in 18 out of 35 samples of fly ash porewaters obtained from Atikokan TGS. Manganese levels ranged from less than 0.02 mg/L to 0.46 mg/L with an average concentration of 0.1 mg/L. Of the eighteen porewaters showing Mn concentrations in excess of PWQO/G, only two were above 0.2 mg/L, one being 0.25 mg/L and the other 0.46 mg/L.

The maximum desirable concentration for Mn (0.05 mg/L), like iron, is aesthetically based. Concentrations related to health effects are much larger than those related to aesthetic considerations. At levels exceeding 0.15 mg/L, some waters may cause undesirable tastes or laundry staining, and, concentrations higher than 0.02 mg/L may encourage the formation of black precipitates in pipes (CCREM, 1987). The current PWQO/G concentration of 0.05 mg/L was set based on the the lowest level that can be reasonably achieved through water treatment (CCREM, 1987). In this respect, it is of interest to note that although Mn concentrations in Canadian drinking water supplies are generally below 0.05 mg/L, levels as high as 4.6 mg/L have been reported (CCREM, 1987). Although elevated Mn concentrations were detected in

the Atikokan fly ash porewaters, manganese concentrations in porewaters generated from bituminous-derived fly ash were well below PWQO/G and would not be of environmental concern.

4.2.6 Phenol:

The maximum desirable concentration (MDC) for phenolic compounds in drinking water is 0.002 mg/L. This value is aesthetically based, that is, the guideline was set primarily for taste and odour considerations, particularly in the case of chlorinated water (MOE, 1983). Concentrations related to health effects are much higher than those related to the aesthetic quality of water. For example, the guideline set by the state of Oklahoma, based solely on health concerns of phenol, is 3.5 mg/L (Barker, 1988).

Phenol concentrations at or exceeding the PWQO/G of 0.002 mg/L were detected in 4 out of 10 porewater samples obtained from the lignite-derived fly ash at Atikokan TGS. Phenol concentrations higher than 0.002 mg/L were not observed in the bituminous fly ash porewaters. The concentrations of phenolic compounds in the Atikokan porewaters ranged from <0.001 mg/L to 0.004 mg/L, with an overall mean of 0.002 mg/L. Of the four samples mentioned, one had a phenol concentration at the PWQO/G level of 0.002 mg/L, two had concentrations of 0.003 mg/L and one sample was found to have a phenol level twice that of the MDC (i.e. 0.004 mg/L).

Since phenol is anionic in nature and has a very high solubility (8.2×10^4 mg/L at 20 ^{o}C), it would be expected to migrate near or at groundwater velocities, experiencing very little, if any, attenuation. However, since elevated phenol concentrations were detected only in a few porewater samples obtained from Atikokan TGS and since these levels were fairly low (the average porewater concentration did not exceed PWQO/G) it is, therefore, not considered to be a potential contaminant causing significant environmental impact.

5.0 SUMMARY

1. Based on the bulk chemical analysis, Ontario Hydro fly ashes are predominantly composed of Si and Al with smaller amounts of Ca, Mg, Na, K and Fe. On a compositional field diagram, the fly ash would be classified as alkaline modic.

2. Fly ash derived from bituminous coal, generally contains higher amounts of trace elements (As, Se, Cr, V and Ti) than fly ash from lignite coal.

3. A comparison of the bulk elemental composition of Ontario Hydro fly ash with the Metropolitan Toronto Regional Conservation Authority and proposed Ontario Ministry of the Environment (MOE) lakefill guidelines suggests that unstabilized fly ash would not be considered a suitable material for lakefilling operations.

4. A review of MOE Regulation 309 leachate extraction results indicates that all of Ontario Hydro fly ashes considered have extract concentrations < 10x the leachate quality criteria outlined in Schedule 4 of Regulation 309. On the basis of these results, Ontario Hydro fly ash would be classified as a non-hazardous (non-registerable, non-leachate toxic) waste in accordance with Regulation 309.

5. Results of the MOE distilled water leach tests show that the distilled water extracts had relatively lower solute concentrations than the Regulation 309 leachate extract. This suggests that a monofill landfill site should be considered as a preferred option for the disposal of Ontario Hydro fly ash in order to reduce the likelihood of acid generation and microbial activity associated with municipal refuse that would increase trace element concentrations in any subsequently generated leachates.

6. A comparison of ash porewater analyses with the Ontario Drinking Water Quality Objectives and Guidelines (PWQO/G) was made to identify solutes which could cause possible groundwater contamination in a landfill scenario. Bituminous fly ash porewaters exhibited alkaline pH's and concentrations of Ca^{2+} and SO_4^{2-} in excess of the PWQO/G and would therefore be considered as potential contaminants depending on the initial background water quality of the site and potential water use. In contrast to the bituminous ash results, porewaters obtained from lignite-derived ash did not exhibit major ion concentrations in excess of the PWQO/G.

7. With respect to minor and trace elements, depending on the thermal generating station, concentrations of B, Se, As, and, to a lesser extent, Cr, Mn and phenol have been detected in fly ash porewaters in excess of the PWQO/G.

8. Cr was detected at levels exceeding the PWQO/G in only a few porewater samples and these values were considered to be anomalous. Cr was therefore not considered to be a potential contaminant of environmental concern.

9. Phenol concentrations at or exceeding the aesthetically-based PWQO/G value of 0.002 mg/L were detected only in some porewaters obtained from lignite-derived fly ash with the overall average concentration of phenolic compounds in the lignite ash porewaters not exceeding the PWQO/G. Similarly, elevated concentrations of Mn were detected in some samples from lignite-derived ash porewaters. However, the maximum desirable concentration for Mn (0.05 mg/L), like iron, is aesthetically based and concentrations related to health are generally much larger than those related to aesthetic consideration. As a result, although Mn concentration in Canadian drinking water supplies are normally <0.05

mg/L, levels as high as 4.5 mg/L have been reported. No phenol or Mn concentrations above the PWQO/G were observed in porewaters taken from bituminous-derived fly ash.

10. Although boron is generally considered to be potentially mobile in groundwater environments, B concentrations in fresh fly ash were shown to decrease to below the Ontario drinking water objectives after being eluted with less than 10 pore volumes of deionized water. Therefore, although initially present at concentrations higher than the PWQO/G value of 5 mg/L, boron levels will decline significantly over time and will cease to be of environmental concern.

11. Results of a long-term column leaching experiment indicate that Se and As are associated with the exterior glassy surface of the ash particle and would persist at concentrations in excess of PWQO/G values for an extended period of time. However, both As and Se levels should eventually decline to levels no longer considered to be an environmental threat to drinking water supplies.

12. Attenuation processes, such as adsorption and precipitation, are significant for both Se and As species in solution. Therefore, both solutes are considered to be fairly immobile in most groundwater environments.

6.0 RECOMMENDATIONS FOR FUTURE RESEARCH

1. At present, very little is known concerning the behaviour and transport of trace elements in a composite system consisting of two geochemically contrasting media such as would occur at a fly ash (alkaline or acidic)/clay liner interface. Therefore, research is needed in the area of geochemical discontinuities and their effect on mass transport.

2. The effect of the addition of stabilizing agents (eg., cement, lime, etc) may have a significant effect on the release of trace elements from fly ash and needs to be evaluated.

3. The determination of the leaching properties of Ontario Hydro coal ash under both leachate recycling and constant renewal conditions requires investigation. In conjunction with this, the maximum amount of leachable constituents relative to the bulk trace element chemistry, under both scenarios, should be determined.

4. Basic research is needed to assess the trace element distribution in Ontario Hydro coal ashes with respect to the glassy versus metallic phases and surface versus subsurface areas. Work is also required to establish the form of the elemental phases both in the solid ash and in the subsequently generated leachates.

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TABLE 1: DESCRIPTION OF COAL TYPES USED

Station	Coal Type	Origin	Ash Handling
Atikokan	lignite	Saskatchewan	dry landfill
Thunder Bay	lignite bituminous	Saskatchewan W. Canadian	dry landfill
Lambton	bituminous	Appalachian Range (Eastern US)	dry landfill
Lakeview	"	"	no permanent on-site storage
Nanticoke	bituminous blend	50% Appalachian 50% W. Canadian	wet lagoon

TABLE 2: BULK ELEMENTAL ANALYSIS (μg/g)

Element	Lakeview	Nanticoke Lagoon	Nanticoke Precipitator	Atikokan	Average of All Fly Ash*
Ag	--	< 0.05	--	--	38 (30%)**
Al	113000-114512	114882-123353	104000	109080	112500 (10%)
As	76.1-89	46-61	50.4	24.2	89 (30%)
B	78	370-420	--	--	217 (25%)
Ba	360-700	--	1500	8600	1635 (35%)
Ca	28857-34000	32786-39357	44428	109566	33990 (30%)
Cd	< 2-2.5	< 1-1	< 2	< 2	2.5 (25%)
Cl	50	50-150	50	50	--
Co	36-37	--	17	< 10	37 (10%)
Cr	137-170	140-230	100	78	137 (25%)
Cu	84-96	72-91	69	74	84 (40%)
F	61	40-120	65	360	98 (15%)
Fe(III)	80671	48164-69472	53400	36861	--
Fe(total)	84147-89300	--	86591	38451	89290 (10%)
Hg	0.05	0.15-0.17	--	--	0.51 (35%)
K	12784	7886-9630	7803	5230	13020 (15%)
Mg	5698	6995-8019	5969	20381	5393 (20%)
Mn	260	310-465	212.5	182.5	266 (15%)
Mo	12	--	11	14	--
Na	3710	3635-5639	4229	54904	4644 (10%)
Ni	99	59-82	66	86	35 (40%)
P	1485	--	1790	2925	--
Pb	37	28-44	33	30	48 (25%)
S	2683	1802-2904	2082	7608	--
Se	5.06	0.8-10	4.06	1.63	14 (45%)
Si	203047	216533-233333	206500	196187	--
Sr	53	930-1099	147	1750	1219 (10%)
Ti	7314	6421-6774	7074	4736	6273 (24%)
Tl	--	2.3-3.7	--	--	--
U	--	7.8-9.9	--	--	9 (35%)
V	170	--	130	88	185 (25%)
Zn	130	--	98	100	328 (25%)

* from Curtis, 1977
**(standard deviation)

TABLE 3: SUMMARY OF TOTAL CONCENTRATIONS OF SELECTED ELEMENTS IN U.S. FLY ASH
(mean and range reported as ug/g solid)

	Total Fly Ash		--- Fly Ash from Coal Types ---					
			Bituminous		Subbituminous		Lignite	
Element	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Al	113000	(46000 - 152000)	--	--	--	--	--	--
As	156.2	(7.7 - 1385)	219	(11 - 1385)	19.1	(8 - 34)	54.4	(21 - 96)
Ba	1880	(251 - 10850)	--	--	--	--	--	--
Ca	62000	(7400 - 223000)	--	--	--	--	--	--
Cd	11.7	(6.4 - 16.9)	11.7	(<5 - 16.9)	< 5	--	< 5	--
Cl	477	(180 - 1190)	--	--	--	--	--	--
Cr	247.3	(37 - 651)	245	(37 - 609)	73	(41 - 108)	284	(<40 - 651)
Cu	185	(44.6 - 1452)	--	--	--	--	--	--
Fe	76000	(25000 - 177000)	--	--	--	--	--	--
K	14300	(3000 - 25300)	--	--	--	--	--	--
Mg	11800	(1600 - 41800)	--	--	--	--	--	--
Mn	357	(44 - 1332)	--	--	--	--	--	--
Mo	43.6	(7.1 - 236)	56	(7 - 236)	16.5	(<4 - 55)	14.1	(8 - 19.7)
Na	9087	(1300 - 62500)	--	--	--	--	--	--
Ni	141	(22.8 - 353)	--	--	--	--	--	--
P	3070	(1100 - 10340)	--	--	--	--	--	--
Pb	170.6	(21.1 - 2120)	--	--	--	--	--	--
S	12643	(1300 - 64400)	--	--	--	--	--	--
Sb	42.5	(11 - 131)	--	--	--	--	--	--
Se	14	(5.5 - 46.9)	12.3	(<5 - 43.5)	14.2	(<5 - 28.1)	18.4	(<5 - 46.9)
Si	209000	(89500 - 275000)	--	--	--	--	--	--
Sn	43.6	(7.9 - 56.4)	--	--	--	--	--	--
Sr	1331	(204 - 6820)	--	--	--	--	--	--
Ti	6644	(1310 - 10100)	--	--	--	--	--	--
U	18.7	(11.1 - 30.4)	--	--	--	--	--	--
V	271.5	(<95 - 652)	290	(99 - 652)	133	(<25 - 292)	209	(<25 - 268)
Zn	449.2	(27 - 2880)	607	(65 - 2880)	148	(27 - 658)	64.7	(25 - 127)

From: Ainsworth C.C., and D. Rai, 1987. Chemical Characterization of Fossil Fuel Combustion Wastes. EPRI EA-5321, Electric Power Research Institute.

TABLE 4: OPEN WATER DISPOSAL GUIDELINES (from MTCRA, 1988)

Parameters Under Strict Control	Guideline (mg/L)
Hg	0.3
PCB's	0.05
Pb	50
Cd	1.0
Oil and Grease	1500
Volatile Solids (loss on ignition @ 600 degrees Celsius)	60 000
Phosphorus (total)	1 000
As	8.0

Parameters to be Assessed on a Case by Case Basis	
Cu	25
Zn	100
Total Kjeldahl Nitrogen	2 000
Cr	25
Fe	10 000
Ni	25
Co	50
Ag	0.5
Cyanide	0.1
Ammonia	100

TABLE 5: LAKEFILL QUALITY GUIDLELINE RESTRICTED LAND USE
(from MTRCA, 1988)

Parameter	Guideline (mg/L)
As	20
Cd	3
Cr (total)	120
Co	25
Cu	100
Fe	35 000
Pb	375
Hg	0.5
Mo	4
Ni	60
Se	2
Zn	500
Sb	8
PCB's	2
V	70
Oil and Grease	5 000

TABLE 6: CLEAN-UP GUIDELINES FOR SOILS (FROM MOE,1989)

Parameter[2]	Criteria for Proposed Land Use[1,3]			
	Agricultural/Residential/ Parkland[4]		Commercial/Industrial	
	Medium & Fine Textured Soils	Course Textured Soils[5]	Medium & Fine Textured Soils	Course Textured Soils[5]
pH (recommended range)	6-8	6-8	6-8	6-8
EC (mS/cm)[7]	2	2	4	4
SAR[8]	5	5	12	12
Arsenic	25	20	50	40
Cadmium	4[1a]	3[1a]	8[1a]	6[1a]
Chromium (VI)	10	8	10	8
Chromium (total)	1000	750	1000	750
Cobalt	50	40	100	80
Copper	200[1b]	150[1b]	300	225
Lead	500[1a]	375[1a]	1000[1a]	750[1a]
Mercury	1[1a]	0.8[1a]	2[1a]	1.5[1a]
Molybdenum	5[1b]	5[1b]	40	40
Nickel	200	150	200	150
Nitrogen (%)	0.5[9]	0.5[9]	0.6[9]	0.6[9]
Oil and Grease (%)	1[6]	1[6]	1[6]	1[6]
Selenium	2[1b]	2[1b]	10	10
Silver	25	20	50	40
Zinc	800	600	800	600

Notes:

1 Clean-up guidelines recommended by the Phytotoxicology Section, Air Resources Branch, Ministry of the Environment. The guidelines are based primarily on phytotoxicity except for ([1a]) based on human health, and ([1b]) based on health of grazing animals.

2 All units are in ppm (µg/g), dry weight, unless otherwise stated.

3 For comparison with these guidelines, analyses for metal and metalloids must be conducted using an approved strong, mixed-acid digestion procedure. Contact the Laboratory Services Branch of MOE if in doubt about acceptable methods.

4 Guidelines have been endorsed by the OMAF/MOE/MOH Sludge Utilization Committee.

5 Defined as greater than 70% sand and less than 17% organic matter.

6 Guideline given is for fresh oil; for weathered oil (min. 2 yr. exposed on site), the guideline is 2%.

7 EC = electrical conductivity (saturation extract).

8 SAR = sodium adsorption ratio.

9 If nitrogen levels exceed the guidelines, the mineralization of the soils should be evaluated. Additions of nitrogen-based fertilizer may be counter-productive.

TABLE 7: MOE WASTE CLASSIFICATION

Constituent	Reg. 309 Schedule 4 (mg/L)	NON - LEACHATE TOXIC WASTE: Non-Registerable (<10 x Sch. 4) (mg/L)	NON - LEACHATE TOXIC WASTE: Registerable (10-100 x Sch. 4) (mg/L)	LEACHATE TOXIC WASTE (>100 x Schedule 4) (mg/L)
Arsenic	0.05	< 0.5	0.5 to 5.0	> 5.0
Barium	1.0	< 10	10 to 100	> 100
Boron	5.0	< 50	50 to 500	> 500
Cadmium	0.005	< 0.05	0.05 to 0.5	> 0.5
Chromium	0.05	< 0.5	0.5 to 5.0	> 5.0
Cyanide	0.2	< 2.0	2.0 to 20	> 20
Fluoride	2.4	< 24	24 to 240	> 240
Lead	0.05	< 0.5	0.5 to 5.0	> 5.0
Mercury	0.001	< 0.01	0.01 to 0.1	> 0.1
Nitrate	10.0	< 100	100 to 1000	>1000
Nitrite	1.0	< 10	10 to 100	> 100
PCB's	0.003	< 0.03	0.03 to 0.3	> 0.3
Selenium	0.01	< 0.1	0.1 to 1.0	> 1.0
Silver	0.05	< 0.5	0.5 to 5.0	> 5.0
Uranium	0.02	< 0.2	0.2 to 2.0	> 2.0

TABLE 8: LAKEVIEW FLY ASH LEACH TEST RESULTS

Constituent	Distilled Water Leach Test (mg/L)	Reg. 309 Leach Test (mg/L)
Arsenic	0.028 - 0.04	0.01 - 0.05
Barium	0.5	0.08 - 0.5
Boron	4.8 - 5.6	12.7 - 15
Cadmium	< 0.0001	0.0051 - 0.035
Chloride	3.9	0.9 - 1.6
Chromium	0.14 - 0.22	0.02 - 0.11
Copper	< 0.05	0.005 - 0.05
Cyanide	0.02	0.01
Fluoride	0.10 - 0.57	0.12 - 1.8
Iron	< 0.05	< 0.02
Lead	< 0.002	< 0.002 - 0.005
Manganese	< 0.05	0.56 - 0.75
Mercury	<0.0002 - 0.00029	< 0.0001 - 0.00032
Nitrate	0.003 - 0.05	0.004 - 0.22
Nitrite	0.003 - 0.007	< 0.003 - 0.008
PCB's	< 0.0002 - 0.001	< 0.0001 - 0.001
Phenol	0.0005	< 0.0005 - 0.005
Selenium	0.03 - 0.054	0.02 - 0.05
Silver	< 0.0005 - 0.001	0.0005 - 0.002
Sulphate	515	570 - 785
Uranium	< 0.001 - 0.03	0.002 - 0.03
Zinc	0.02	0.08 - 0.37
Radium-226	0.018 Bq/L	0.0026 Bq/L
pH	-	5.1

TABLE 9: NANTICOKE FLY ASH LEACH TEST RESULTS

Constituent	Distilled Water Leach Test (lagoon ash) (mg/L)	Distilled Water Leach Test (precipitator ash) (mg/L)	Reg. 309 Leach Test (precipitator ash) (mg/L)
Arsenic	0.023 - 0.30	0.007 - .015	0.022 - 0.166
Barium	< 1.0	< 0.5	< 1.0 - 5.0
Boron	1.4 - 1.9	2.5 - 6.0	12.3 - 23.4
Cadmium	< 0.0001	< 0.002	< 0.005
Chloride	0.55 - 1.5	-	< 0.5
Chromium	< 0.05	< 0.1	< 0.02
Copper	< 0.001	< 0.05	< 0.047 - 0.06
Cyanide	< 0.04	-	< 0.02
Fluoride	< 0.02 - 0.25	0.074 - 0.91	0.15 - 0.36
Iron	< 0.05	< 0.05	< 0.05
Lead	< 0.0002	< 0.1	< 0.002
Manganese	< 0.05	< 0.05	0.58 - 0.79
Mercury	< 0.0002	< 0.0002	< 0.0002
Nitrate	< 0.005	< 0.1	< 0.002
Nitrite	< 0.005 - 0.009	< 0.15	< 0.003
PCB's	< 0.0001	< 0.03	< 0.0001
Phenol	< 0.001	-	< 0.005
Selenium	< 0.019 - 0.035	0.023 - 0.055	0.024 - 0.09
Silver	< 0.0002	< 0.025	0.0022
Sulphate	40 - 45	21	180 - 330
Uranium	< 0.01	0.0003 - 0.0005	0.0027 - 0.047
Zinc	< 0.02	< 0.02	0.05 - 0.24
TDS	150 - 225	-	-
Radium-226	0.0037 Bq/L	-	-
pH	10.7 - 10.8	11.2	4.9 - 5.2

TABLE 10: THUNDER BAY FLY ASH LEACH TEST RESULTS

Constituent	Distilled Water Leach Test (mg/L)	Reg. 309 Leach Test (mg/L)
Arsenic	0.001	0.002
Barium	< 5	< 1
Boron	7.5	21.1
Cadmium	< 0.02	< 0.02
Chromium	< 0.2	< 0.1
Cyanide	< 0.001	< 0.001
Fluoride	0.13	0.63
Lead	< 0.1	< 0.1
Mercury	< 0.0002	< 0.0002
Nitrite	< 0.06	< 0.06
PCB's	< 0.03	< 0.03
Phenol	< 0.0001	--
Selenium	0.12	0.10
Silver	< 0.0002	< 0.0002
Uranium	< 0.05	< 0.05
pH	10.1	5.2

TABLE 11: ATIKOKAN FLY ASH LEACH TEST RESULTS

Constituent	Reg. 309 Leach Test (mg/L)
Arsenic	0.002
Barium	< 1
Boron	29
Cadmium	0.0045
Chloride	2
Chromium	0.06
Copper	0.02
Cyanide	< 0.02
Fluoride	2.2
Iron	0.59
Lead	< 0.002
Manganese	0.39
Mercury	< 0.0002
Nitrate	0.14
Nitrite	< 0.003
PCB's	< 0.0001
Phenol	< 0.005
Selenium	0.062
Silver	0.002
Sulphate	685
Uranium	0.06
Zinc	0.16
pH	5.2

TABLE 12: LAMBTON FLY ASH LEACH TEST RESULTS

Constituent	Distilled Water Leach Test (mg/L)
Arsenic	0.01 - 0.014
Barium	< 1.0
Boron	2.4 - 3.4
Cadmium	0.0001
Chloride	0.83 - 3.8
Chromium	< 0.05
Copper	< 0.001
Cyanide	0.04
Fluoride	0.1 - 0.3
Iron	< 0.05
Lead	<0.002
Manganese	< 0.05
Mercury	< 0.0002
Nitrate	0.008 - 0.031
Nitrite	0.003 - 0.008
PCB's	0.001
Phenol	0.001
Selenium	0.015 - 0.02
Silver	< 0.0002
Sulphate	155 - 405
Uranium	< 0.01
Zinc	< 0.02
TDS	328 - 640
Radium-226	< 0.0037 Bq/L
pH	9.5 - 10.1

TABLE 13: LAKEVIEW FLY ASH POREWATER ANALYSIS

Constituent (includes # of samples)	Porewater (mg/L) Mean	Range
Aluminum (7)	1.78	(0.6 - 3.6)
Arsenic (5)	0.044	(0.012 - 0.067)
Barium (2)	< 0.03	(< 0.03)
Boron (6)	28.6	(17.8 - 51)
Bromide (5)	1.92	(< 0.1 - 4.3)
Cadmium (6)	< 0.0001	(< 0.0001)
Calcium (7)	540	(460 - 640)
Chloride (7)	78.6	(34 - 141)
Chromium (5)	0.08	(< 0.02 - 0.2)
Copper (6)	0.003	(< 0.002 - 0.005)
Cyanide (1)	< 0.005	(< 0.005)
Iron (5)	0.07	(< 0.02 - 0.15)
Lead (6)	0.004	(< 0.002 - 0.017)
Magnesium (7)	4.7	(1.32 - 10)
Manganese (2)	< 0.01	(< 0.01)
Mercury (6)	0.0001	(< 0.0001 - 0.0002)
Molybdenum (7)	7.4	(0.5 - 14)
Nitrate (4)	7.4	(0.01 - 15.4)
Nitrite (1)	0.17	(0.17)
Phenol (1)	< 0.0005	(< 0.0005)
Potassium (7)	170	(29 - 260)
Selenium (5)	0.028	(0.012 - 0.052)
Silver (5)	0.0005	(< 0.0002 - 0.0006)
Sodium (7)	350	(31 - 550)
Strontium (7)	11.8	(8.5 - 23)
Sulphate (7)	2007	(1450 - 2400)
Uranium (5)	< 0.005	(< 0.005)
Vanadium (7)	0.11	(0.05 - 0.29)
Zinc (6)	0.02	(< 0.01 - 0.04)
TDS (2)	3065	(1980 - 4150)
Radium-226 (3)	< 0.0037	< 0.0037 Bq/L
pH	--	8.6 - 9.5

TABLE 14: LAMBTON FLY ASH POREWATER ANALYSIS

Constituent (includes # of samples)	Porewater (mg/L) Mean	Range
Antimony (1)	< 0.000005	(< 0.000005)
Arsenic (3)	0.037	(0.012 - 0.055)
Beryllium (1)	< 0.0002	(< 0.0002)
Cadmium (3)	0.0004	(0.0003 - 0.0005)
Calcium (3)	602	(580 - 625)
Cesium (1)	< 0.000001	(< 0.000001)
Chloride (3)	48	(17 - 92)
Chromium (1)	< 0.0001	(< 0.0001)
Cobalt (1)	< 0.000005	(< 0.000005)
Copper (3)	0.0167	(0.014 - 0.020)
Lead (3)	0.0027	(0.002 - 0.003)
Magnesium (3)	29.6	(10.3 - 60)
Manganese (3)	0.000017	(0.00001 - 0.00003)
Potassium (3)	56	(37 - 87)
Selenium (3)	0.010	(0.008 - 0.013)
Silver (3)	0.0005	(<0.0005 - 0.0008)
Sodium (3)	92	(43 - 148)
Strontium (6)	12.8	(11 - 14)
Sulphate (3)	1743	(1547 - 1909)
Zinc (3)	0.023	(0.02 - 0.03)
pH	--	9.05 - 9.60

TABLE 15: NANTICOKE FLY ASH POREWATER ANALYSIS

Constituent (includes # of samples)	Porewater (mg/L) Mean	Range
Arsenic (12)	0.012	(< 0.002 - 0.025)
Boron (12)	3.62	(0.23 - 13.2)
Calcium (12)	141	(54 - 265)
Chloride (12)	26	(14 - 45.8)
Fluoride (4)	0.24	(< 0.1 - 0.43)
Iron (4)	< 0.05	(< 0.05)
Magnesium (12)	0.66	(0.045 - 3.7)
Manganese (8)	< 0.02	(< 0.02)
Nitrate (4)	< 1.0	(< 1.0)
Potassium (12)	29.4	(3.5 - 71.6)
Selenium (12)	0.01	(< 0.002 - 0.028)
Silicon (12)	6.3	(1.5 - 12.9)
Sodium (12)	52.7	(9.4 - 103)
Strontium (12)	6.3	(0.86 - 17)
Sulphate (12)	294	(18 - 590)
TDS (8)	666	(194 - 1200)
pH	--	7.9 - 11.8

TABLE 16: ATIKOKAN FLY ASH POREWATER ANALYSIS

Constituent (includes # of samples)		Porewater (mg/L) Mean	Range
Arsenic	(35)	< 0.001	(< 0.001)
Barium	(35)	< 0.5	(< 0.5 - 1.8)
Beryllium	(35)	< 0.0002	(< 0.0002)
Boron	(35)	0.06	(< 0.02 - 0.1)
Cadmium	(35)	0.00015	(< 0.0001 - 0.0008)
Calcium	(34)	31.9	(6.2 - 47)
Chloride	(35)	4.5	(0.5 - 67)
Chromium	(35)	< 0.05	(< 0.05 - 0.1)
Copper	(35)	0.007	(0.001 - 0.025)
Cyanide	(11)	< 0.04	(< 0.04)
Fluoride	(12)	0.2	(0.1 - 0.41)
Iron	(35)	< 0.05	(< 0.05 - 0.86)
Lead	(35)	< 0.002	(< 0.002)
Magnesium	(35)	2.76	(0.1 - 4.1)
Manganese	(35)	0.10	(< 0.02 - 0.46)
Mercury	(35)	< 0.0002	(< 0.0002 - 0.0008)
Molybdenum	(35)	< 0.5	(< 0.5)
Nickel	(35)	< 0.005	(< 0.005 - 0.01)
Nitrate	(35)	1.28	(< 0.004 - 4.52)
Nitrite	(12)	0.03	(< 0.003 - 0.11)
PCB's	(8)	< 0.0002	(< 0.0002)
Phenol	(10)	< 0.002	(< 0.001 - 0.004)
Potassium	(35)	1.0	(0.21 - 1.9)
Selenium	(35)	< 0.001	(< 0.001)
Silicon	(28)	7.13	(2.8 - 9.91)
Silver	(35)	< 0.0002	(< 0.0002)
Sodium	(35)	3.44	(1.5 - 15)
Strontium	(35)	0.04	(< 0.02 - 0.14)
Sulphate	(35)	6.03	(3.0 - 9.4)
Titanium	(35)	< 0.5	(< 0.5)
Uranium	(12)	< 0.01	(< 0.01 - 0.02)
Vanadium	(35)	< 0.005	(< 0.005)
Zinc	(35)	< 0.02	(< 0.02 - 0.04)
TDS	(11)	137	(106 - 160)
pH		--	6.2 - 8.1

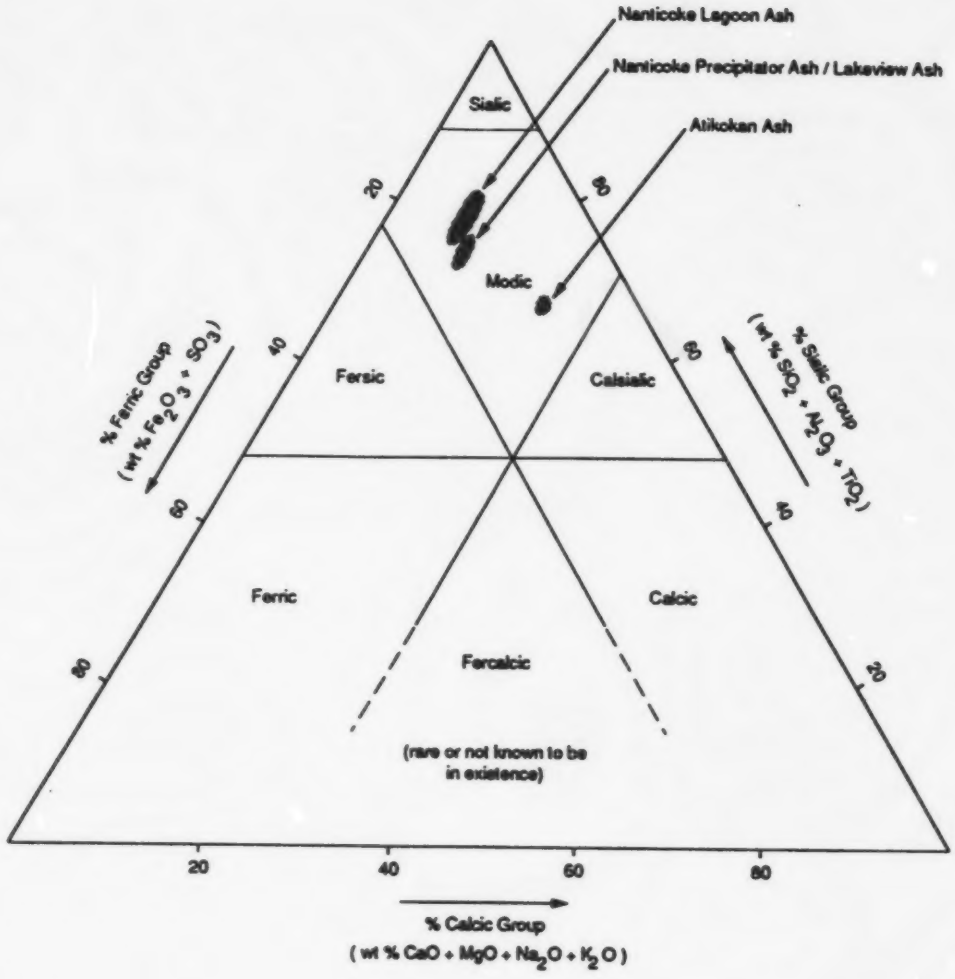
TABLE 17: ONTARIO DRINKING WATER OBJECTIVES

Constituent	Maximum Acceptable Concentration (mg/L)
Arsenic	0.05
Barium	1.0
Boron	5.0
Cadmium	0.005
Chromium	0.05
Cyanide	0.2
Fluoride	2.4
Lead	0.05
Mercury	0.001
Nitrate	10.0
Nitrite	1.0
PCB's	0.003
Selenium	0.01
Silver	0.05
Uranium	0.02
Radium-226	0.1 Bq/L
	Maximum Desirable Concentration (mg/L)
Chloride	250
Copper	1.0
Iron	0.3
Manganese	0.05
Phenols	0.002
Sulphate	500
Zinc	5.0
Total dissolved solids	500

TABLE 18

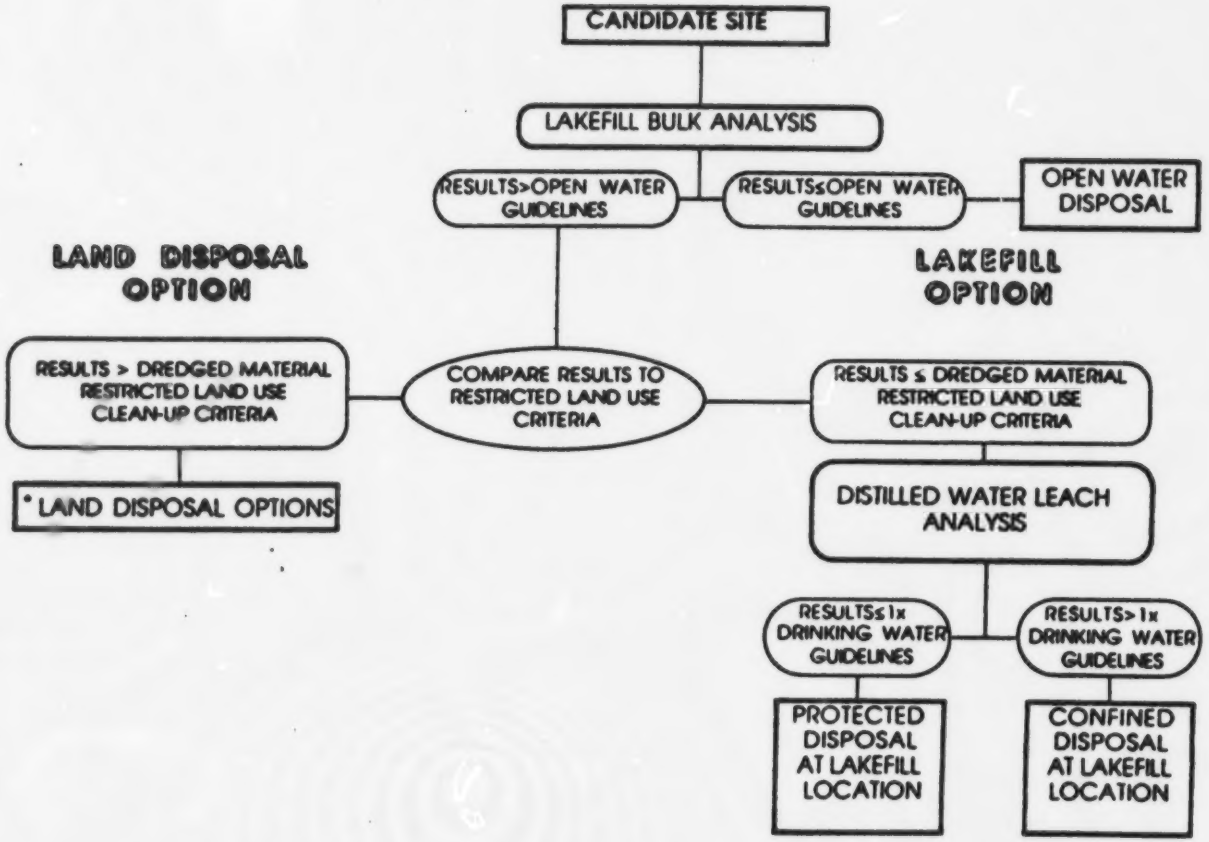
CALCULATED KD VALUES FOR SELENIUM ($mL.g^{-1}$) (from TICKNOR et al. 1988)

Redox Condition:	Aerobic			Anaerobic		
	Ionic Strength					
	SCSS			SCSS		
Solid	1%	10%	100%	1%	10%	100%
Calcite	6.6	2.5	1.3	7.5	3.4	3.7
Quartz	1.6	1.3	0	8.5	1.5	1.4
Goethite	34	31	40	56	36	46
Hematite	17	31	84	79	71	89
Chlorite	52	28	27	77	35	21
Illite-Muscovite	37	23	31	56	35	32
Kaolinite	17	20	28	53	43	43
Gypsum	11	11	4.4	9.7	9.2	7.5
Muscovite	12	8.3	2.2	11	5.5	5.9
Epidote	1.6	1.3	0	8.5	1.5	1.4
Granite	7.9	7.1	3.7	43	27	28
Gabbro	7.9	7.1	7.5	31	22	26
Basalt	11	14	16	31	20	36



COAL FLY ASH CLASSIFICATION SCHEME

FIGURE 1


CANDIDATE SITE
LAKEFILL BULK ANALYSIS
RESULTS>OPEN WATER GUIDELINES
RESULTS≤OPEN WATER GUIDELINES
OPEN WATER DISPOSAL
LAND DISPOSAL OPTION
LAKEFILL OPTION
RESULTS > DREDGED MATERIAL RESTRICTED LAND USE CLEAN-UP CRITERIA
COMPARE RESULTS TO RESTRICTED LAND USE CRITERIA
RESULTS ≤ DREDGED MATERIAL RESTRICTED LAND USE CLEAN-UP CRITERIA
* LAND DISPOSAL OPTIONS
DISTILLED WATER LEACH ANALYSIS
RESULTS≤1x DRINKING WATER GUIDELINES
RESULTS>1x DRINKING WATER GUIDELINES
PROTECTED DISPOSAL AT LAKEFILL LOCATION
CONFINED DISPOSAL AT LAKEFILL LOCATION

* Subject to Ontario Regulation 309
and/or M.O.E. Decommissioning Guidelines

FIGURE 2: FLOW CHART FOR IDENTIFYING DISPOSAL OPTIONS USING THE IMPROVED LAKEFILL QUALITY CONTROL PROGRAM (FROM MTRCA, 1988)

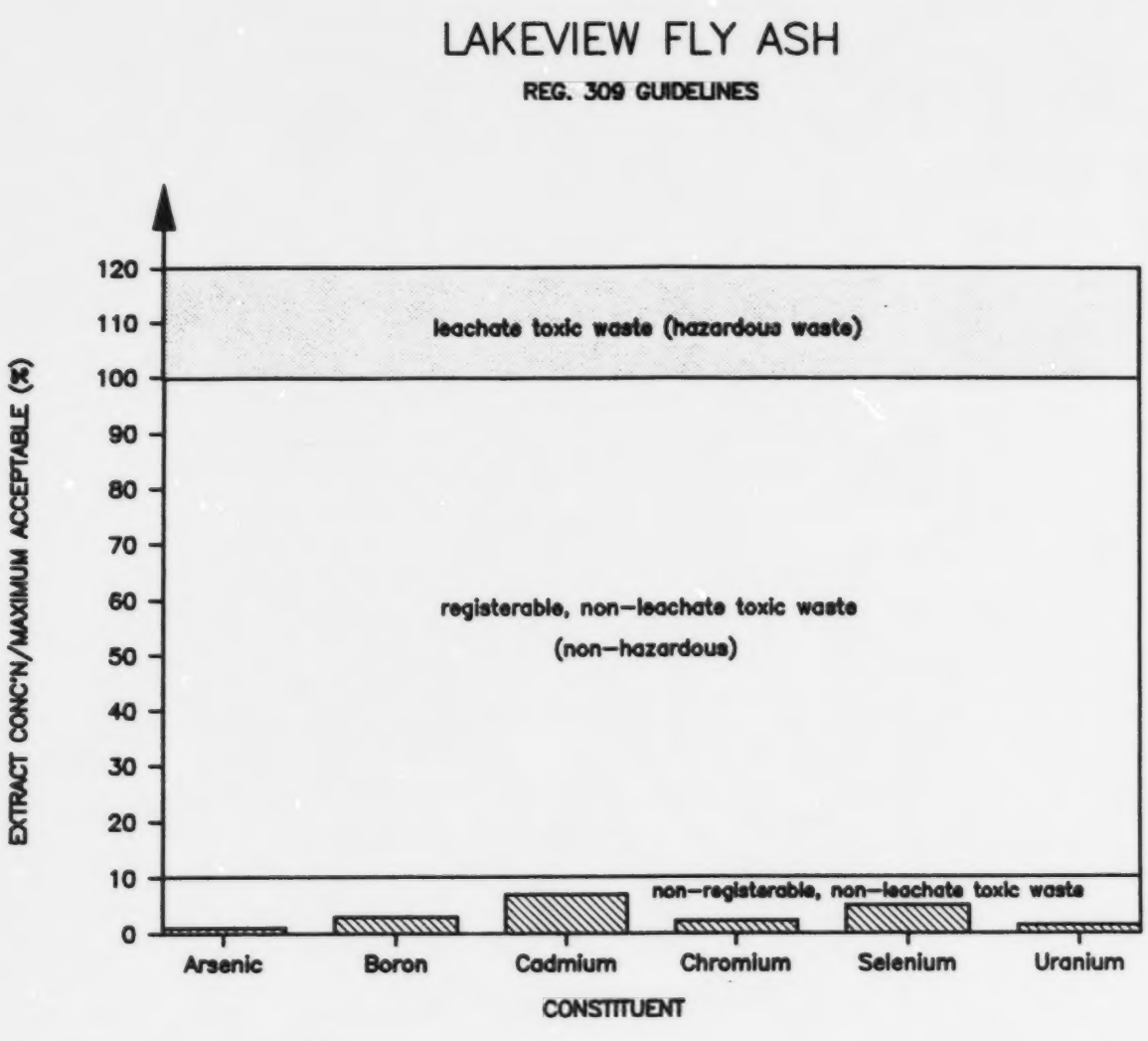


FIGURE 3: LAKEVIEW FLY ASH EXTRACT/REG. 309

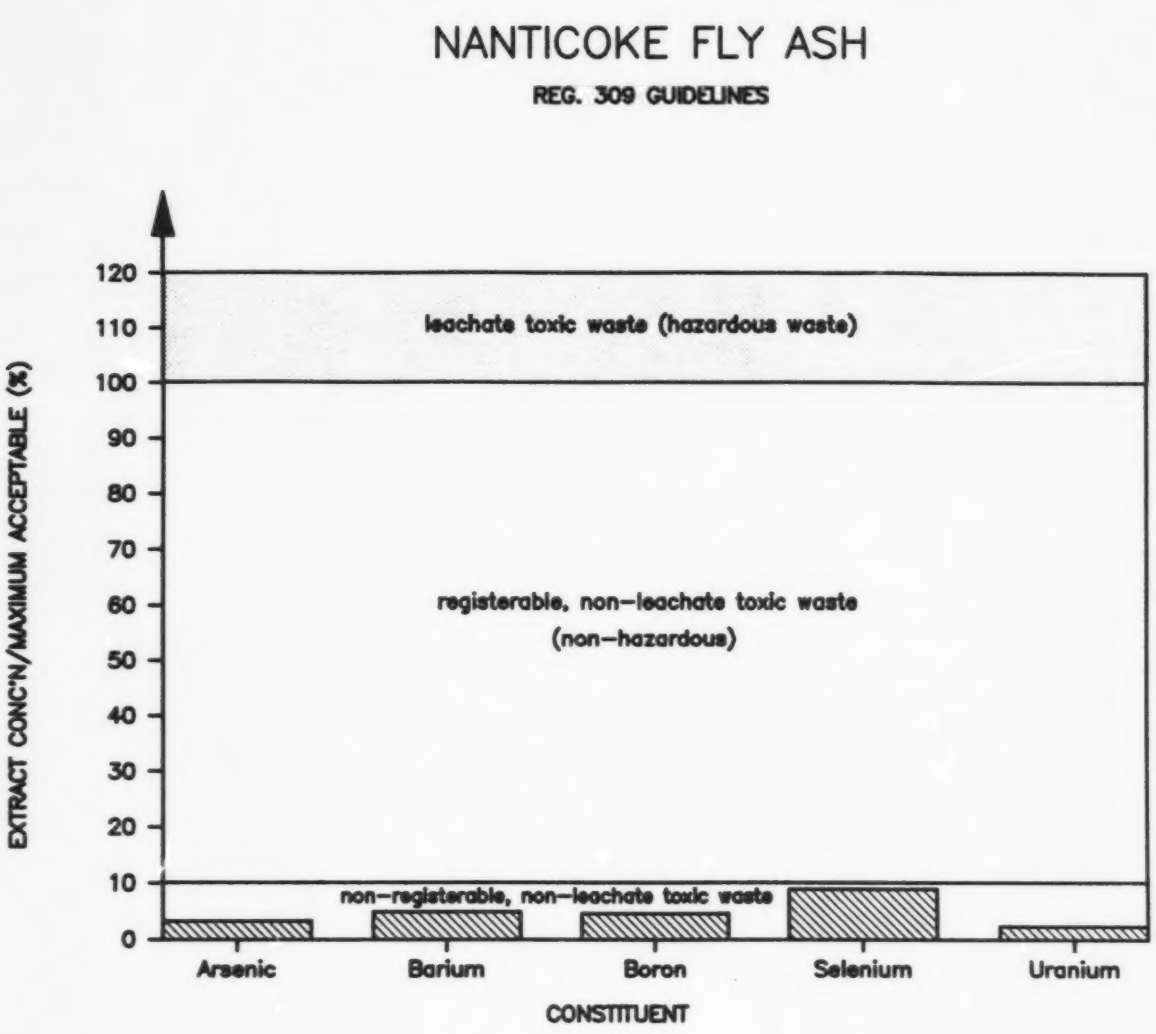


FIGURE 4: NANTICOKE FLY ASH EXTRACT/REG. 309

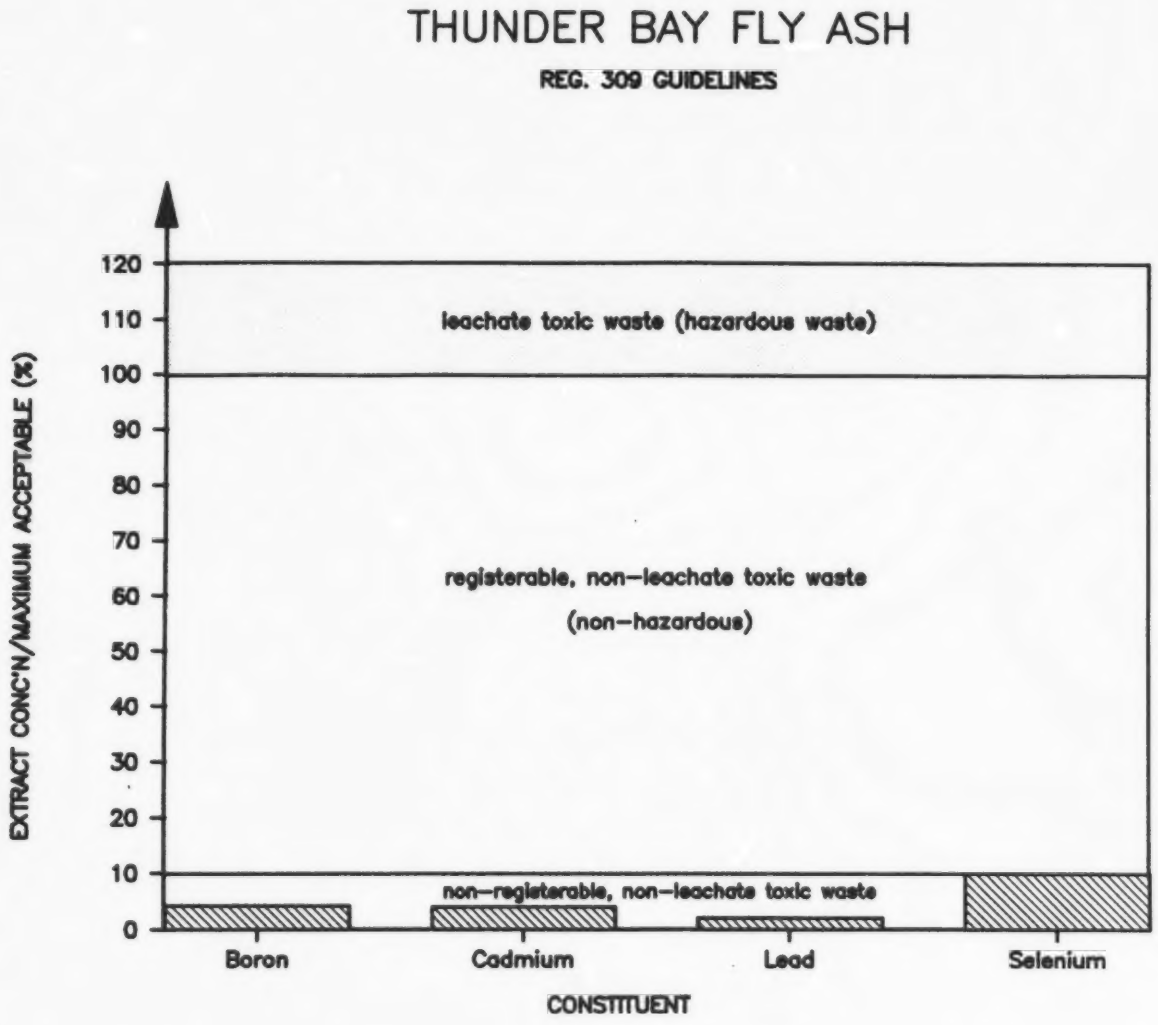


FIGURE 5: THUNDER BAY FLY ASH EXTRACT/REG. 309

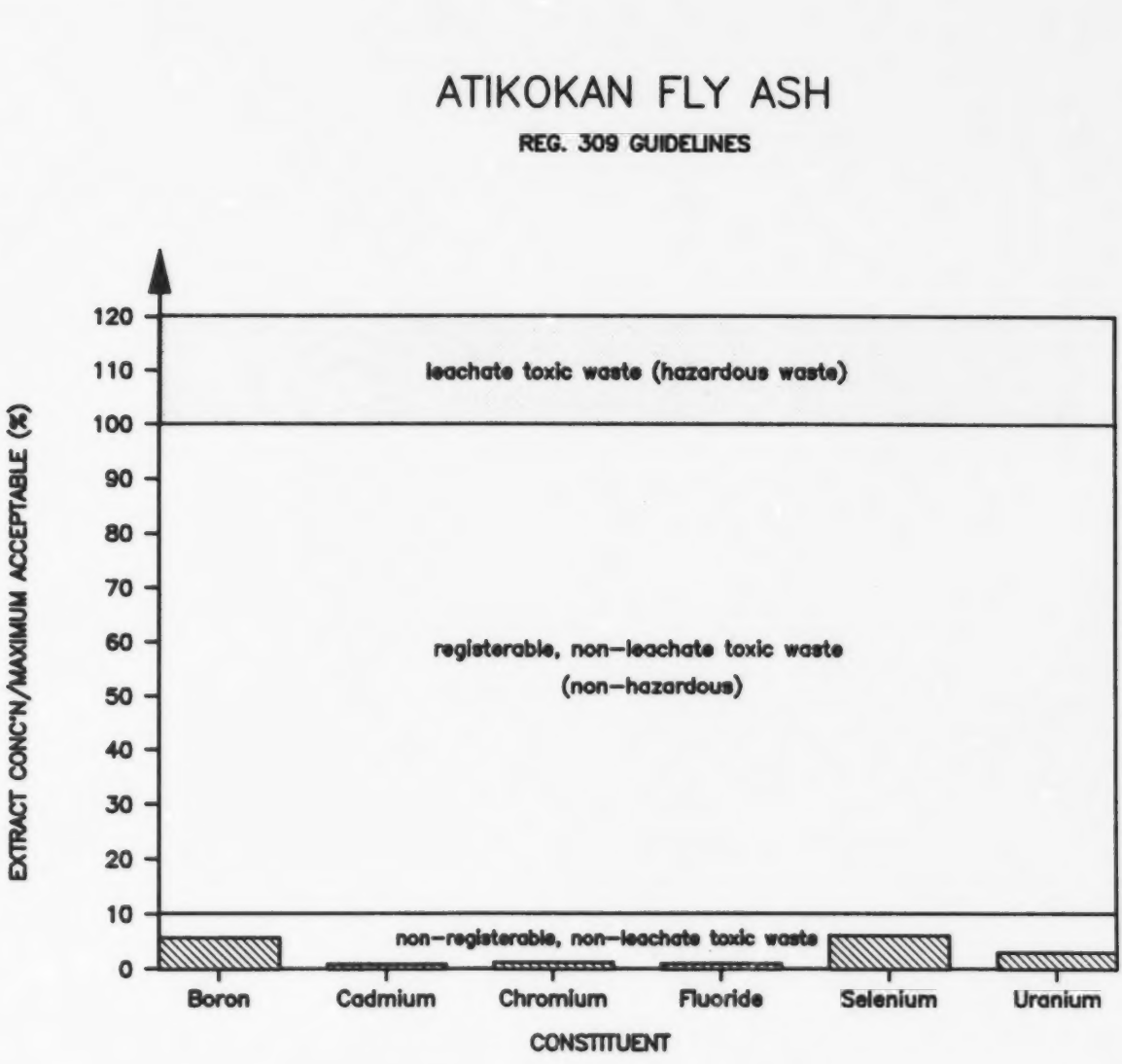


FIGURE 6: ATIKOKAN FLY ASH EXTRACT/REG.309

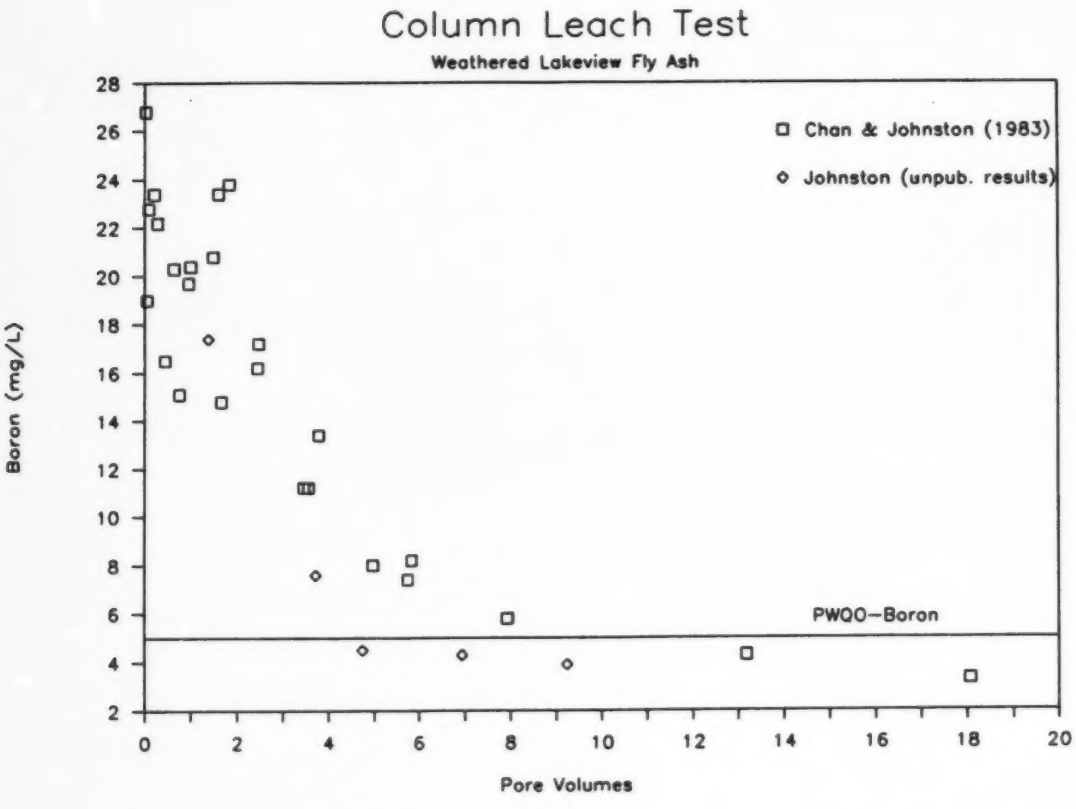


FIGURE 7: SHORT-TERM COLUMN: EXPERIMENTAL RESULTS FOR BORON

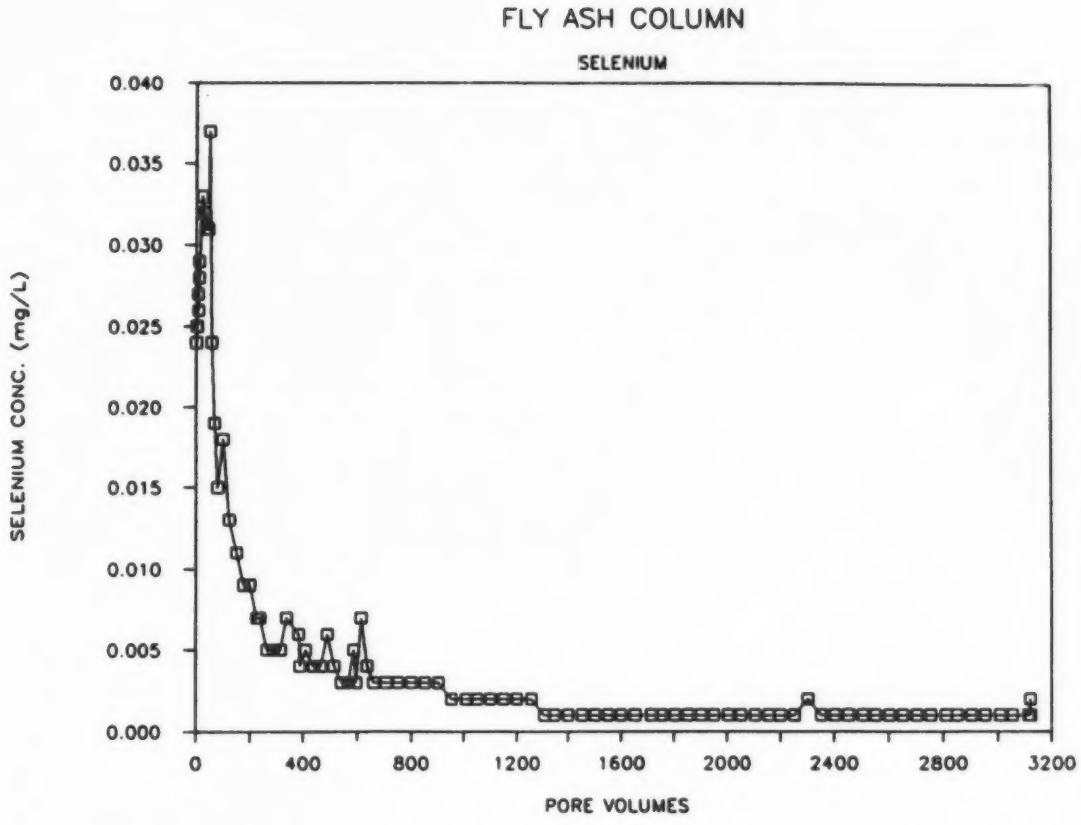


FIGURE 8: LONG-TERM COLUMN: EXPERIMENTAL RESULTS FOR SELENIUM

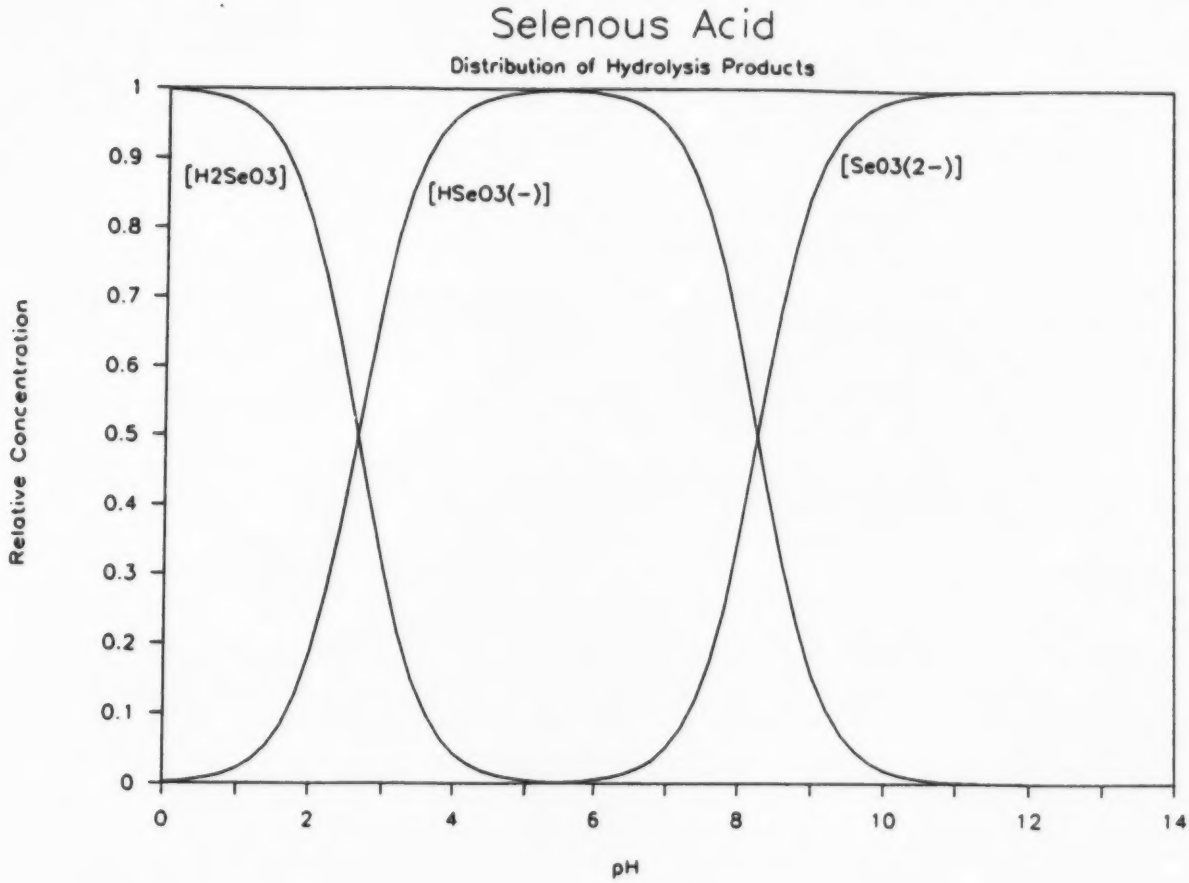


FIGURE 9: DISTRIBUTION OF HYDROLYSIS PRODUCTS: SELENOUS ACID

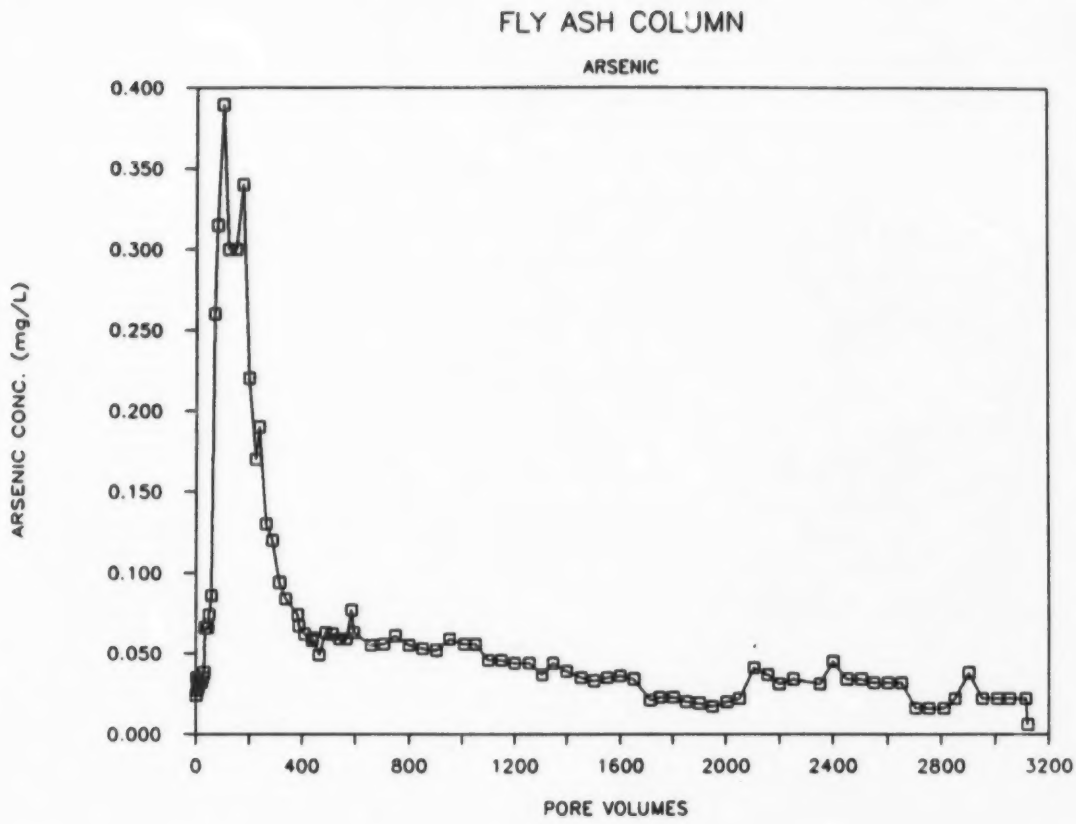


FIGURE 10: LONG-TERM COLUMN: EXPERIMENTAL RESULTS FOR ARSENIC

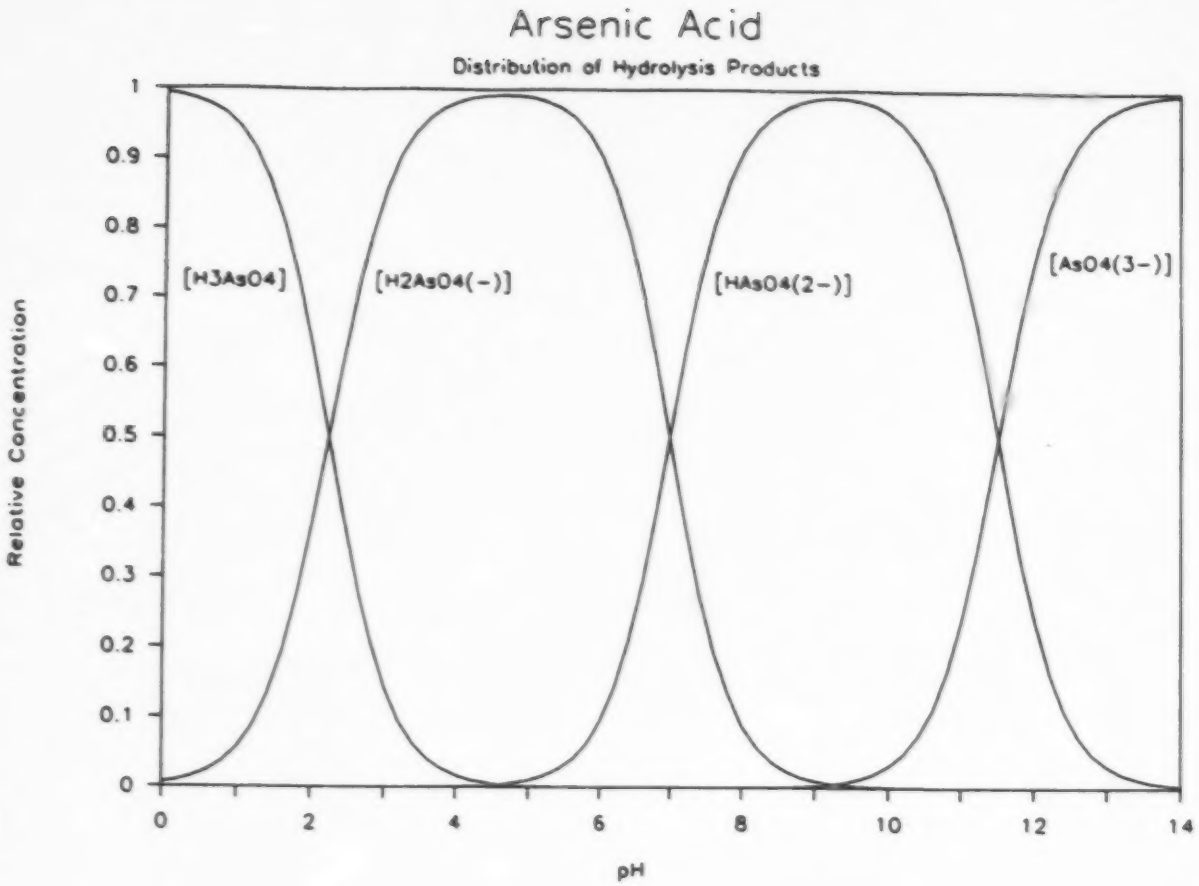


FIGURE 11: DISTRIBUTION OF HYDROLYSIS PRODUCTS: ARSENIC ACID